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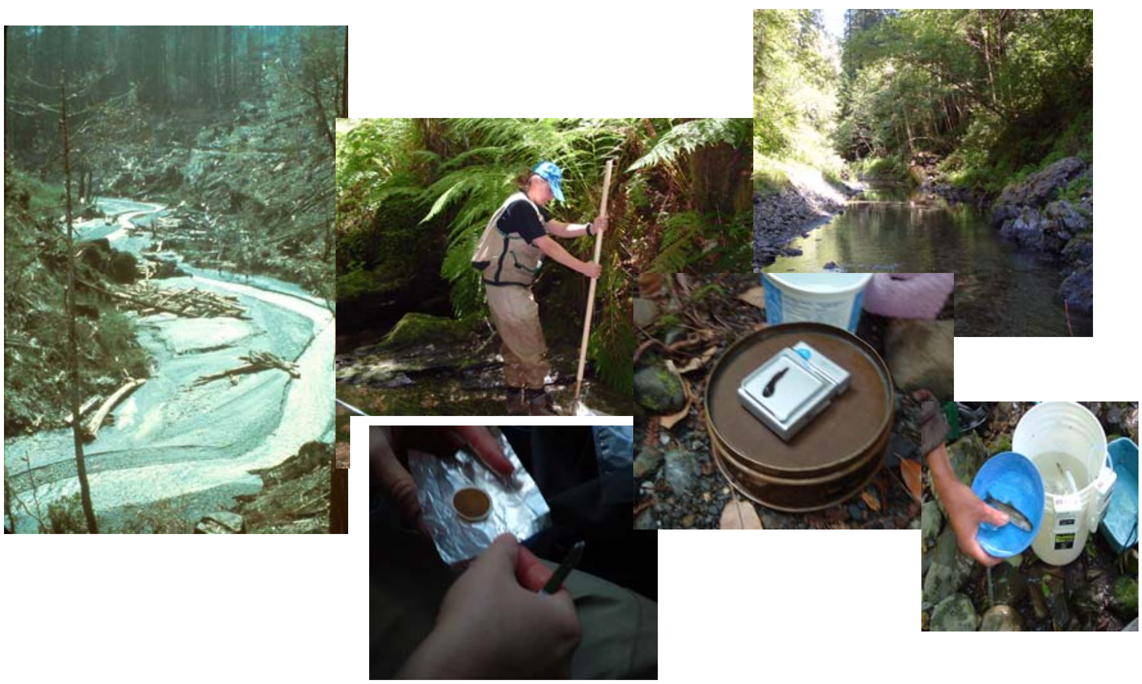
Assessing Changes in Stream Health Following Watershed Restoration: A 30-Year Perspective, Redwood Creek Basin, Humboldt County, California



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Objectives

Between 1973 and 1975, the U.S. Geological Survey (USGS) collected aquatic biological data at 50 sites in the Redwood Creek watershed. At that time the watershed had extensive areas of timber harvest, road construction, unstable hillslopes and eroding stream channels. The National Park Service initiated a watershed restoration program in lower Redwood Creek in 1978. Since then basin conditions have changed drastically through removal of about 300 km of abandoned logging roads and revegetation of previously logged areas. The purpose of this retrospective study was to assess the impacts of upslope restoration efforts on stream health by comparing conditions in the 1970's to those in 2004 and 2005.

Methods

Tributary stream reaches at several of the USGS study sites (Iwatsubo and others, 1975) were resampled in both the spring and fall for two years, in 2004 and 2005 using the same techniques as in the 1970's. Results were compared to those from 30 years ago to determine if there was an improvement in aquatic health as measured by abundance and diversity of benthic macroinvertebrates, periphyton, stream amphibians, fish condition, and physical habitat parameters. Additional sites with recent road restoration work were also sampled to provide a baseline for future studies.

Physical habitat

Physical conditions in the streams have improved since the 1970's. Sediment transport rates in Redwood Creek have decreased since the 1970's. A prominent change in physical habitat in the sampled basins was an increase in riparian canopy since 1974. Much of the timber harvest at that time did not include stream buffers, and so long reaches of stream were exposed to direct sunlight. During the last 30 years, the riparian zones in these impacted reaches have become revegetated, primarily with alders. In addition, several tributary channels aggraded during floods in 1972 and 1975. Flood-deposited sediment has now been flushed out of these tributaries, and only small remnants of flood terraces remain. Bridge Creek showed the most dramatic channel degradation, with about 3 m of chanel bed lowering since 1975. Also, the percentage of channel length in pools and maximum pool depths both increased in Bridge Creek. Anecdotal evidence suggests a coarsening of channel bed substrate since the 1970's, but quantification of bed material particle size was not done at the study sites in the earlier period. Revegetation of riparian zones and transport of flood deposits occurred in tributaries both with and without road removal work.

Although no summer temperature data are available for the sample sites in the 1970's, current temperature conditions are adequate for salmonids. No tributaries exceeded a Maximum Weekly Average Temperature Value (MWAT) of 16.8°C, the level of concern for juvenile coho (Welsh and others, 2001). Water temperatures reached their summer maxima in late July or early August for all years of record. Stream temperatures were highest in Bridge Creek, which has the largest drainage area and is farthest inland of all the study sites.

Benthic Macroinvertebrates

The USGS used Surber samplers to collect benthic macroinvertebrates in the 1970's, so similar sampling was conducted in this study. In addition, to establish a baseline for future work according to current standards of the California State Bioassessment Protocol, kick sampling was also conducted. Only riffles were sampled. An Index of Biological Integrity (IBI) was calculated for all the sites. There was high variability among the sites and pristine sites did not necessarily have the highest IBI ratings. There was also high temporal variability, between seasons and between years. Large floods in 1975 and 1997 and erosional disturbances (landslides, road failures, and adjustment of restored stream crossings) probably influenced the abundance and diversity of invertebrates, but there was not a clear trend in bioassessment metrics and the degree of disturbance.

Although there were not clear trends in all the bioassessment metrics, some generalizations can be made which suggest improved stream health since the 1970's. Filtering-collectors filter fine particulate matter and are expected to increase in response to disturbance (Harrington and Born, 2000). Percentages of filtering collectors were higher in the spring of 1974 than in 2004 at most sites. Gathering collectors are the macrobenthos that collect or gather fine particulate matter and are expected to increase in response to disturbance (Harrington and Born, 2000). Late summer percentages of gathering collectors tended to be higher in 1975 than in 2004. The ratio of filtering collectors (FG) to gathering collectors (GC) tended to be higher in streams that were mostly pristine or were not recently disturbed. The diversity index values for spring sampling were higher in 2004 than in the 1970's at most sites. Based on the ratio of scrapers to shredders and total collectors, most of the sampled streams were heterotrophic (they are dependent upon allochthonous organic inputs more than autochthonous primary production). Insects exhibiting 2+ year life cycles can be indicative of more stable channel conditions. In general, long-lived species were more abundant in 2004-2005 than in 1974.

Periphyton

Periphyton growth rates measured in 2004-2005 were lower in almost every stream than those measured in the early 1970's. This was probably due to the increase in canopy cover at the sampling sites. Canopy cover consists of both hardwoods (alder) and conifers (redwood and Douglas fir), but by late spring and summer both types of vegetation are effective in shading these small streams. Water discharge also affects periphyton growth. Higher spring flows in 2005 than in 2004 probably account for the lower spring accrual rates at most sites in 2005. Differences in upstream land use, including the degree of road removal work, were not apparent in periphyton growth rates.

Amphibians

Stream reaches in undisturbed redwood forests had significantly higher biomass and density of tailed frogs than streams in basins with various degrees of road removal work. There was not a strong difference in streams with recent restoration work as compared to streams with older (1980's) restoration work. This result is consistent with other studies showing that recovery of headwaters amphibian assemblages may be suppressed for many decades after timber harvest, even after recovery of the forest canopy.

Fish

Sample sizes of fish captured or detected in the sampled tributary streams were small, so our results must be considered preliminary. Preliminary results suggest an improvement in fish condition and distribution since 1974. Coho salmon were not present in Bridge Creek, one of the largest tributaries of Redwood Creek, in the 1974-75 surveys, but were found there in both 2004 and 2005. In Tom McDonald Creek and Little Lost Man Creek at the bridge, coho were present in 1974 but were not found in 1975, and they were detected in both streams in 2004 and 2005.

Fish condition was defined by the relationship between length and weight of steelhead. Based on a small sample size of captured fish, steelhead condition improved in Harry Weir and Bridge Creeks from 1974 to 2005. There was no significant difference in steelhead condition in Little Lost Man Creek, a pristine site, or at Redwood Creek in Redwood Valley, upstream of park boundaries in a managed landscape.

Further Research

'Disturbance' in the tributaries of Redwood Creek within Redwood National and State Parks results from a combination of past timber harvest and road construction, large floods and landslides, and recent road decommissioning work. Because 'disturbance' cannot be stated in terms of a single factor, indices of biotic integrity, based on periphyton, benthic macroinvertebrates, amphibians and fish, do not follow a simple pattern of lowered integrity with increased disturbance. Landscape-scale disturbance characteristics may not be reflected in the reach-scale sampling design used in this study. Another constraint of this retrospective study is that the original study sites were not randomly selected and treatments were not conducted within an experimental framework. To more effectively assess the impact of upland road removal on stream health, an experimental framework with more control over treatments and non-treatments would be needed. Elam Creek, a sub-basin that will have roads removed in the near future, could provide such an opportunity.

An underlying assumption of the design of this study was that pristine sites would exhibit different characteristics than sites impacted by past timber harvest and road removal activity. Nevertheless, during the study it became clear that within the group of pristine streams, variability was high. In 2003, a group of scientists representing many disciplines engaged in a two-day field reconnaissance to visit these pristine sites, and the collective interpretation was that it is actually this variability that gives those sites less modified by human activities much of their resilience. In this view, parameters in pristine sites could exhibit high variance at any point in time; human disturbance could lead to a reduction in variance. As a consequence, reference sites might actually be more variable than disturbed ones (Lisle and others, in press), and indices of biological integrity may be expected to vary as well. The trajectories of stream recovery following disturbance vary according to the type and magnitude of disturbance and the physical conditions of the specific watershed. More research is needed on the linkage of physical and biological processes and on understanding conditions in a watershed context.

Acknowledgements

This project was conducted cooperatively by the United States Geological Survey (Redwood Field Station, Arcata, California) and National Park Service (Arcata, California). This research was funded primarily by the California Department of Fish and Game. We appreciate discussions and suggestions provided by Dr. Ken Cummins, Dr. Walt Duffy, David Anderson, Baker Holden, Michael Sparkman, and Jon Lee. Nik Erickson was an enthusiastic field assistant. Rick Iwatsubo kindly let us review his original field notes and photographs from the 1970's study.

Chapter 1 Introduction

Background

Redwood National Park, California, was established in October 1968 to preserve significant examples of primeval coastal redwood (*Sequoia sempervirens*) forests and the associated streams and forests. In the 1970's the Redwood Creek basin was undergoing extensive timber harvest. Concerns over upstream logging and road construction on park resources prompted federal legislation expanding Redwood National Park in 1978 by 19,425 ha, and a 14,569 ha park protection zone upstream of park boundaries was also added. Much of this land was privately owned before being acquired by the National Park Service and commercial timber harvest and road construction resulted in landslides, gullies, channel aggradation, increased stream temperature, and changes in nutrient inputs.

As part of the expansion legislation (PL 95-250) Congress authorized a program of watershed rehabilitation for the newly acquired lands. The National Park Service initiated a watershed restoration program in 1978. The long-term goal of the land rehabilitation program is to reduce sedimentation from past logging, to return the downstream portion of the Redwood Creek basin to a reasonable facsimile of the natural state and to protect irreplaceable park resources. Since the beginning of the program in 1978, basin conditions have changed drastically through removal of about 300 km of abandoned logging roads and revegetation of previously logged areas. However, about 190 km of abandoned roads within the parks still need to be treated. Redwood Creek is currently listed as sediment and temperature impaired under the Clean Water Act, section 303 (d) and water quality issues within the creek continue to be an item of concern for park managers. Aquatic habitats within the creek have and continue to be altered by erosion and sedimentation due to floods and land use, but watershed restoration efforts within the parks are attempting to improve stream conditions. An assessment of the effectiveness of watershed restoration on stream health is a necessary component of adaptive management in order to guide future watershed restoration work.

Many anadromous fish streams in coastal California have been damaged by various land use activities, including timber harvest and road construction. Coho salmon (*Oncorhynchus kisutch*), chinook salmon (*Oncorhynchus tshawytscha*) and steelhead trout (*Oncorhynchus mykiss*) populations are in serious decline in many north coastal streams. Loss of habitat from erosional and sedimentation problems is a major contributing cause of this decline (Meehan, 1991; Spence and others, 1996). The chinook, coho and steelhead trout populations of Redwood Creek are still listed as threatened under The Endangered Species Act. Rapid and substantial declines in anadromous fish habitat and fish populations in the Redwood Creek basin prompted Nehlsen and others (1991) to classify the basin’s fall chinook salmon stocks, coho salmon stocks and cutthroat trout stocks as being at moderate risk of extinction. In addition, tailed - frogs (*Ascaphus truei*) and torrent salamanders (*Rhyacotriton olympicus*), both listed as species of concern by the State of California, are present within the Redwood Creek basin.

Between 1973 and 1975, the U.S. Geological Survey (USGS) collected aquatic biological data at 50 sites in the Redwood Creek watershed. Benthic macroinvertebrates, periphyton and fish were sampled. At that time, steelhead trout captured in the Redwood Creek drainage basin were substantially slimmer than the steelhead trout population representative of small California coastal streams (Averett and Iwatsubo, 1995). Sites from the USGS study (Iwatsubo and others, 1975) were resampled in both the spring and fall for two years, in 2004 and 2005 (Figure 1-1, Appendix 1-1 – 1-4). Results were compared to those from 30 years ago to determine if there was an improvement in aquatic health as measured by abundance and diversity of benthic macroinvertebrates, periphyton accrual rates, stream amphibians, fish condition, and physical habitat parameters.

The purpose of this retrospective study was to assess the impacts of upslope restoration efforts on stream health through a comparison of current conditions with those documented in the 1970's. Restoration on a watershed scale is a relatively new science, and it is important to gage the efficacy of past efforts to guide future work. Water quality assessment, based on the abundance and diversity of key indicator organisms and physical measurements of habitat, can be used to guide management decisions to preserve and aid in the recovery of the park's aquatic biota. The use of biological indicators for water quality assessment (both sediment and temperature) is well established. Sediment loads and disturbances play a key role in the formation of benthic macroinvertebrate community structure. Macroinvertebrates are commonly used to test environmental conditions of water resources (Plafkin and others 1989). Benthic macroinvertebrates are the major prey item for aquatic amphibians and fish, and so are a critical component of aquatic ecosystems that needs to be monitored. Amphibians are also used as a vertebrate indicator to assess the health of local aquatic conditions (Vitt and others 1990) because they display limited dispersal, as opposed to migratory salmonids. Both the tailed frog and southern torrent salamanders appear to be sensitive to timber harvest and sedimentation (Corn & Bury, 1989) and can be indicators of suitable stream conditions.

Additional benefits from the project include public education opportunities resulting from the update of the aquatic biological data base for the parks. Voucher specimens of benthic macroinvertebrates developed during the sampling have been delivered to the curator of Redwood National and State Parks, and these can be used as a interpretive as well as a research tool. Benthic macroinvertebrate sampling and identification from 2004 to 2005, using the California Stream Bioassessment Protocol, will form a baseline for future monitoring of watershed conditions in the Redwood Creek basin.

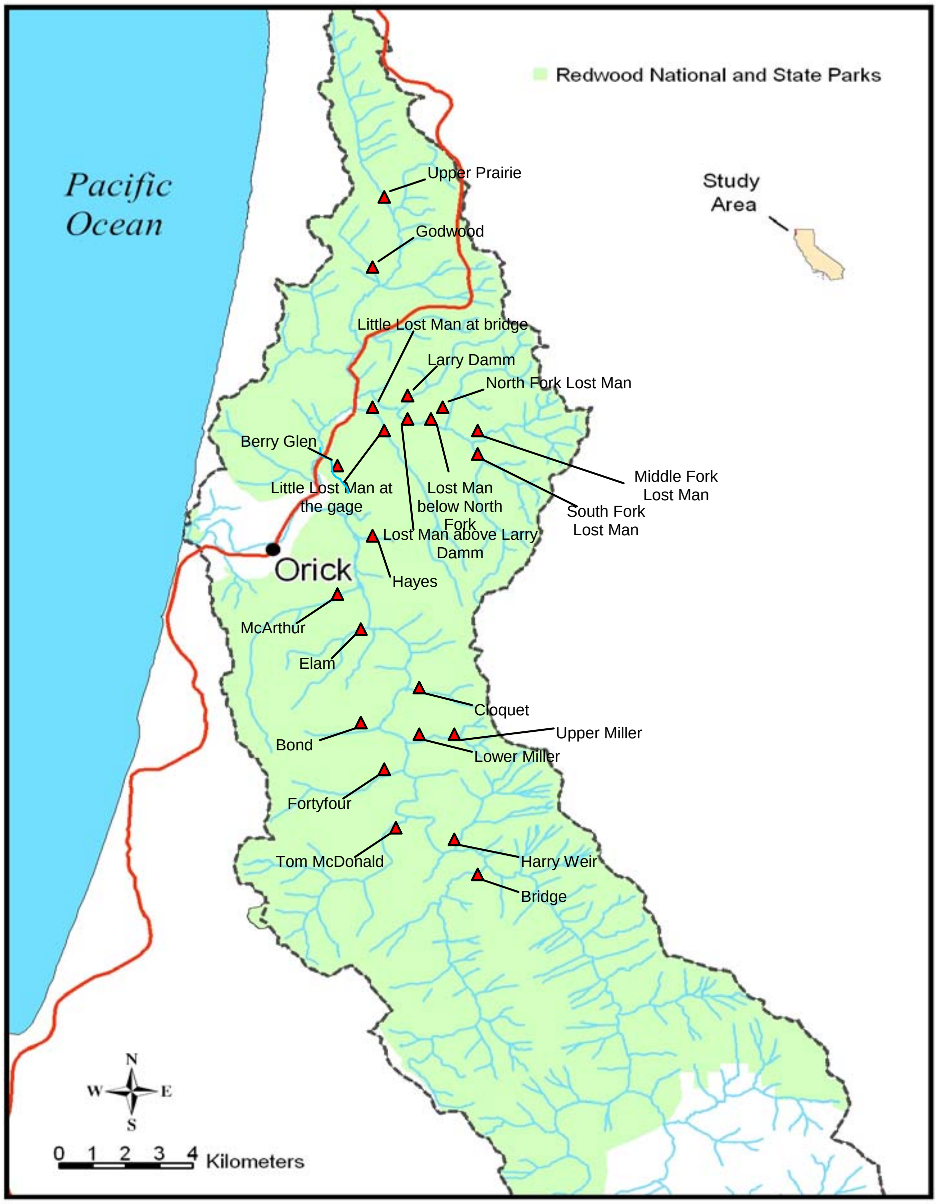


Figure 1-1. Location map of lower Redwood Creek basin showing sampling sites in tributary basins**.**

Previous Studies

Road decommissioning or removal has been used in watershed restoration programs since the 1970's. The goal of such work is to reduce sediment input to streams from road-related erosion problems, which may lead to improved conditions for fish and other aquatic biota (Harr and Nichols, 1993). Several studies have addressed the effectiveness of road removal work in terms of sediment production over a several-year time period (Klein, 1987; Bloom, 1998; Madej, 2001a; Pacific Watershed Associates, 2005). Klein (2004) also investigated the storm-by-storm erosional response at excavated stream crossings in terms of elevated turbidity. Nevertheless, few studies have documented biological response to upland road restoration work. The degree of revegetation and invasion by exotic vegetation in excavated stream channels was documented by Madej and others (2000), and Switalski and others (2004) propose other biological monitoring that should accompany road decommissioning work. Duffy (in review) provides specific suggestions to monitor the biological impact of upland restoration work; however, implementation of such monitoring is still in its infancy.

Description of Watershed Restoration Work

The focus of the watershed restoration work in Redwood National and State Parks is on the removal of abandoned logging roads. Forest roads are significant sources of sediment (Megahan and Kidd, 1972; Janda and others, 1975; Best and others, 1995). Road cuts and drainage structures, such as culverts, can disrupt natural drainage patterns. Stream crossings fail when culverts plug with sediment or wood, or are too small to convey storm discharge. In these cases, the road fill at the stream crossing may be removed by erosion. Drainage structures can divert streams out of their natural course onto unchannelled hillslopes when the structures fail to function properly, causing gullies. Road cuts can intercept groundwater and increase the amount of surface runoff (Wemple, 1998). In addition, widespread surface runoff from the road bench and cutbanks flows into inboard ditches, which commonly deliver fine sediment to channels.

Typical road treatments include decompacting the road surface, removing drainage structures, excavating road fill from stream channels and exhuming the original streambed and streambanks, excavating unstable sidecast fill from the downslope side of road benches or landings, and filling in or draining the inboard ditch. There has been an evolution in treatment styles since 1978. Treatments in the early 1980's decompacted the road surface and constructed drains perpendicular to the road alignment to dewater the inboard ditch. Following this treatment the roads were mulched with straw, seeded and replanted with native vegetation (Figure 1-2). As the National Park Service program progressed, park geologists began to use more intensive treatment methods, which included partially outsloping the road surface by excavating fill from the outboard edge of the road and placing the material in the inboard ditch at the based of the cutbank. This technique required more earth-moving. By the 1990's geologists commonly prescribed complete recontouring of the road bench (total outslope), in which the cutbank was covered by excavated fill, original topsoil from the outboard edge of the road was replaced on the road bench were possible, stream channels were excavated to the original channel bed elevation, streambanks were extensively reshaped and the road bench was fully recontoured (Figure 1-3). Total outsloping involves moving a great deal of road fill.

a

c

b

d

Figure 1-2. Typical stream channel excavation in the 1980's. a) Abandoned logging road with intact culvert before treatment. b) Immediately following stream crossing excavation. In this case, rock armor and check dams were installed on the channel bed to prevent downcutting. c) Less than one year later, revegetation of the streambanks is well underway. d) Three years after treatment, alders have revegetated most of the ground disturbed during treatment.

a

b

Figure 1-3. An example of more intensive road removal. a) Abandoned logging road before treatment. b) The road bench is obliterated and the hillslope is recontoured. Stumps uncovered during excavation indicate the location and elevation of the original hillslopes.

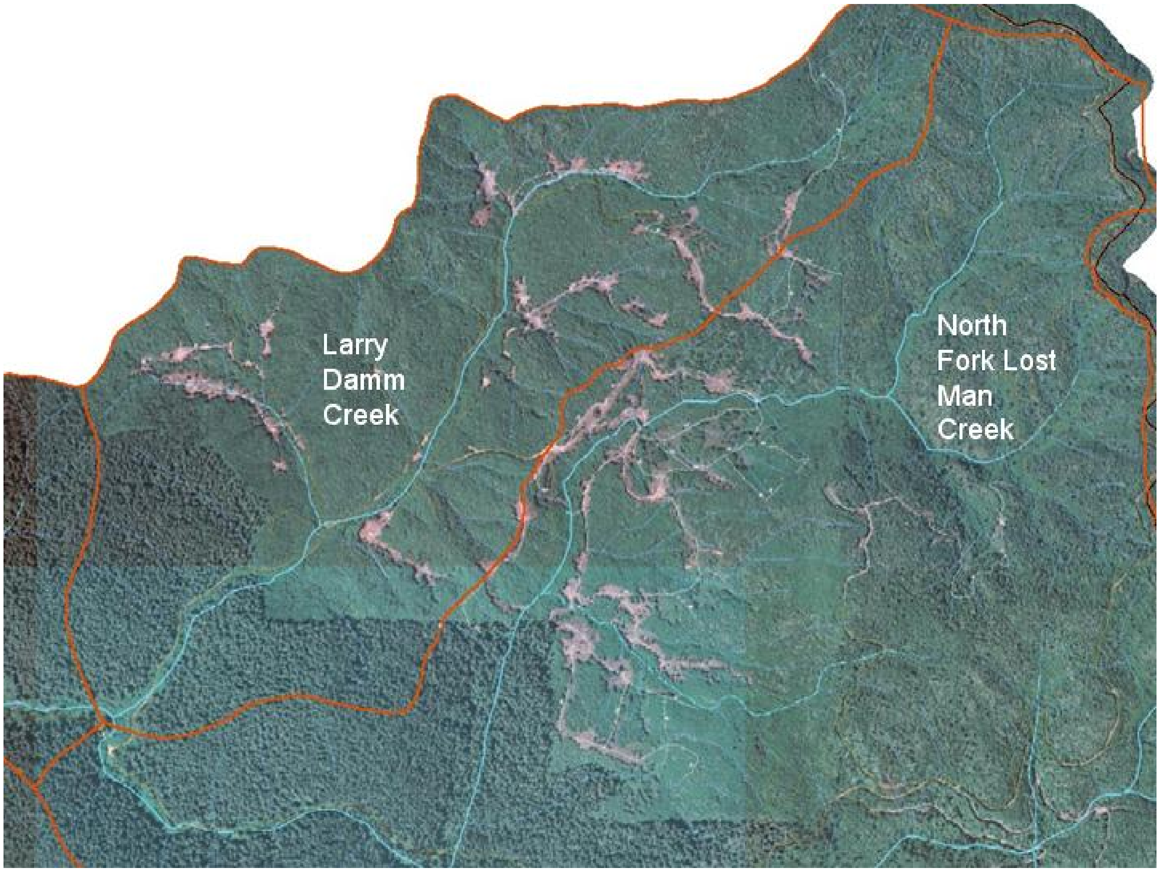


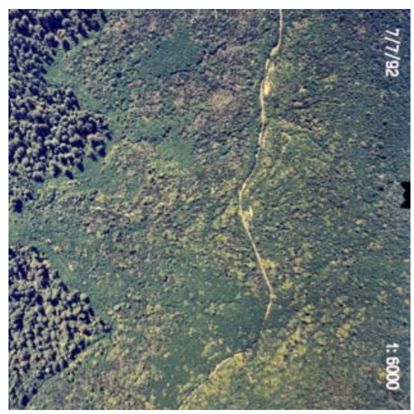
Figure 1-4. Recent road removal work in the Lost Man Creek watershed is evident in aerial photographs by the removal of trees and exposure of bare ground during the heavy equipment phase of restoration.

Figure 1-4 shows the extent of recent ground disturbance associated with road removal work in two sub-basins of the Lost Man Creek watershed. Channel armoring was seldom used in this phase, but trees felled during road treatment were later placed in the stream channels and on the treated road surface. On some road segments, excavated road fill was removed from the road bench and transported to a more stable location (export outslope).

Because road removal involves heavy equipment work and the removal of vegetation, there is commonly a short-term increase in erosion at the site until the newly excavated streambanks settle and become revegetated. It is presently unknown how this short-term flush of sediment affects aquatic biota near the excavation sites. Harris (2005) lists ways to monitor road removal activities.

The above descriptions refer to active restoration through road removal. In effect, there has been another type of restoration in Redwood National and State Parks, that of passive restoration. Since 1978 logging and road construction have ceased on park lands, and previously logged terrain is revegetating naturally. Although abandoned logging roads still pose an erosional threat on these lands, the amount of bare ground has decreased as alders and other early successional species revegetate the hillslopes (Figure 1-5). Elam Creek, for example, is a sub-basin that has had no active road removal but has had more than 25 years of regrowth on the clearcut logging blocks and along stream channels. The relative effectiveness of passive restoration in restoring stream health as opposed to active restoration is not known.

1978



1992

Figure 1-5. Bond Creek basin in 1978 showing an extensive network of skidtrails and disturbed headwater channels. The same area of Bond Creek in 1992 showing extensive regrowth of young alder on previously logged hillslopes.

Chapter 2 Study Sites and Physical Habitat Characteristics

Description of Study Sites

Redwood Creek drains a 720 km^2 watershed in the mountainous, coastal region of northern California. It is one of several major streams and rivers that flow from the Klamath Mountains and Coast Ranges into the Pacific Ocean. The creek originates near elevations of 1500 m and flows approximately 80 km in a narrow, elongated drainage basin north-northwest to the Pacific Ocean near Orick, California. Its many tributaries are generally short and steep.

The climate is predominantly maritime with warm, dry summers and cool, wet winters. The Redwood Creek drainage basin receives an average of 137cm of rainfall per year, as measured at the Orick rain gage, with the majority of the rain occurring between November and May. The dominant riparian vegetation consists of redwood (*Sequoia sempervirens,* Douglas fir (*Pseudotsuga menziesii*), red alder (*Alnus rubra*), tan oak (*Lithocarpus densilorus*) and bigleaf maple (*Acer macrophyllum*). Understories are mixtures of elderberry (*Sambucus racemosa*), salmonberry (*Rubus spectabilis*), blackberry (*Rubus ursinus*), thimbleberry (*rubus parviflorus*), stink currant (*Ribes bracteosum*) and ferns. The most common vertebrates at all sites included steelhead salmon (*Oncorhynchus mykiss*), coho salmon (*Oncorhynchus kisutch*), Pacific giant salamanders (*Dicamptodon tenebrosus*) and tailed frogs (*Ascaphus truei*).

None of the basins has undergone timber harvest since the expansion of Redwood National Park in 1978, and formerly clearcut and select cut areas have been allowed to revegetate naturally. The basins have had varying degrees of road removal work done. Many of the basins had a moderate amount of road removal done in the 1980's, but Elam and Berry Glen Creeks have not had any road removal work (Table 2-1). Since 2001 road removal work has been focused primarily in the Larry Damm and Lost Man Creek basins, and the degree of road crossing excavations has been more extensive than at sites restored in the 1980's (Table 2-1).

Disturbance can be defined in several ways, so it is difficult to rank the basins according to a single criterion. 'Percent old growth' is inversely related to the amount of past timber harvest in the basin, and is commonly used as a ranking factor. Four basins have almost a complete cover of old growth redwood forests, whereas five basins have less than 10 percent old growth remaining (Table 2-1). Nevertheless, the value can be misleading. For example, the Upper Miller Creek study stream reach is located in a pocket of old growth, even though only 5 percent old growth remains in the basin as a whole, and this study site had the greatest volume of in-channel large wood of all sampled streams. Road density has also been used as a disturbance ranking factor, but in the Redwood Creek basin there are two types of relevant road densities. Table 2-1 lists both the road density of existing roads and road density of restored roads. Road removal work may cause a short-term increase in sediment yield as the newly excavated stream crossings adjust during the first few winter high flow seasons. The extent of these adjustments is related to size and extent of excavations (Madej, 2001a), so excavation volume data are also listed in Table 2-1. Finally, a 12-year return period storm in

Table 2-1. Road data for selected watersheds of Redwood National and State Parks. Basins are sorted by volume of road fill excavated from road-stream crossings during road removal work on a per unit drainage area basis.

Sub-Basin	Existing Road Density (km/km^2)	Density of Removed Roads (km/km^2)	Volume excavated from crossings m^3/km^2	Date of Last Disturbance*	1997 Landslide density (m^3/km^2)	Dominant Bedrock**
Godwood	0.00	0.00	0	n/a	0	PPpc
Upper Prairie	1.05	0.00	0	1940	0	PPpc
Hayes	0.71	0.00	0	n/a	0	Kjfc
Little Lost Man at gage	0.40	0.00	0	n/a	0	Kjfl
Little Lost Man at bridge	0.40	0.00	0	1972	300	Kjfl
Elam	3.31	0.00	0	1978	0	Kjfr
Berry Glen	1.19	0.00	0	1965	0	Kjfr/Kjfc
Cloquet	1.21	0.81	1180	1982	0	Kjfr/Kjfc
Harry Weir	1.34	1.28	3170	1985	0	Kjfr/Kjfc
Lower Miller	1.80	1.18	3240	1988	4000	Kjfr
Fortyfour	2.40	1.37	4600	1997	900	Kjfr
Upper Miller	2.02	0.90	4620	1997	3850	Kjfc
Bond	1.59	1.27	4740	1982	1700	Kjfr
Bridge	1.13	1.88	5370	1997	2300	Kjfr
Middle Fork Lost Man	2.58	1.03	5540	2001	0	Kjfl/ PPpc
South Fork Lost Man	2.64	0.40	6270	2001	0	Kjfl
McArthur	2.71	0.96	7770	2000	2800	Kjfr
Lost Man above Larry Damm	2.20	0.70	10520	2001	700	Kjfl/ PPpc
Lost Man below North Fork	2.30	0.77	11440	2001	750	Kjfl/ PPpc
Tom McDonald	0.94	1.58	16270	2001	400	Kjfr

Table 2-1 continued.

Sub-Basin	Existing Road Density (km/km^2)	Density of Removed Roads (km/km^2)	Volume excavated from crossings m^3/km^2	Date of Last Disturbance*	1997 Landslide density (m^3/km^2)	Dominant Bedrock**
North Fork Lost Man	1.77	2.31	25740	2001	2400	PPpc/Kjfl
Larry Damm	0.41	2.62	48940	2003	100	PPpc

*Disturbance is a major road removal project, large debris torrent or landslide, highway construction or logging.
** See text for lithologic descriptions.

January, 1997, initiated about 360 landslides in the Redwood Creek basin. In basins which were affected by large debris torrents or landslides, the landslide volume delivered to the stream channel, normalized by drainage area, is also listed on Table 2-1. The distance between the site of disturbance (stream crossing excavation or landslide) and the study reach is different from sub-basin to sub-basin.

We used a cluster analysis to group sub-basins with similar characterics together. The variables used to categorize sub-basins were: percent old growth forest, existing road density, density of removed roads, 1997 landslide density, and volume excavated from stream crossings per km^2 (Table 2-1). To form the clusters, the procedure begins with each sub-basin in a separate group. It then combines the two closest sub-basins to form a new group, and repeats the procedure until only one group remains (Appendix 2- 1). This procedure separated out basins that are mostly pristine, moderately disturbed, and disturbed by recent road work or landslides. Results of the biological monitoring described throughout this report are displayed by sub-basins clusterd in these relative disturbance categories.

Timing of disturbance also varies. Figure 2-1 shows the cumulative volume of road fill excavated from stream crossings during road removal work in three watersheds, and exemplifies the range of conditions found in Redwood National and State Parks. Road restoration work commenced in the Bridge Creek basin in 1979, and little work has been done since 1990. In contrast, Larry Damm Creek had no restoration work until 2002, but extensive restoration work occurred within a few year period. The Tom McDonald Creek basin has had the greatest volume of road fill removed from stream crossings, but the work has been spread across almost two decades. We would expect that physical and biological responses would differ according to the timing as well as the

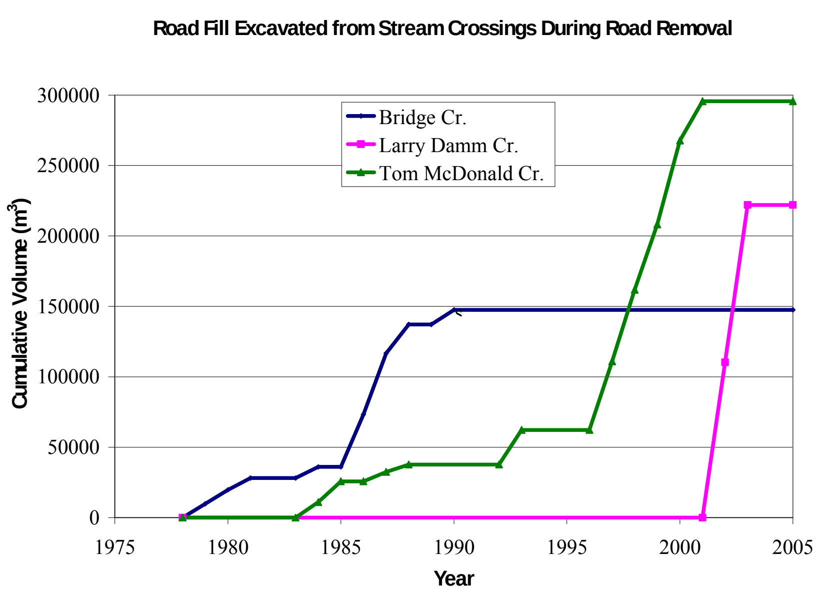


Figure 2-1. The cumulative volume of road fill excavated from stream crossings during road removal work in selected tributaries of Redwood Creek.

magnitude of restoration activities. However, because this is a retrospective study based on site selections made in the 1970's, we were unable to control for all types of channel disturbances in the Redwood Creek watershed.

The streams in this study drain a variety of terrain, and the bedrock geology affects the particle size of sediment entering the streams and the dominant erosional processes in a watershed. The Franciscan Assemblage underlies most of the study area. A full description of the bedrock geology is given by Cashman and others (1995). The dominant types of bedrock found in this study area are:

Kjfc: Incoherent Unit of Coyote Creek. Interbedded sandstones and mudstones.
Kjfl: Coherent unit of Lacks Creek. Massive sandstones with some mudstones.
Kjfr: Schist of Redwood Creek. A quartz-mica schist commonly associated with deep, red and clay-rich regolith.
PPpc – Prairie Creek Formation –weakly consolidated shallow marine and fluvial sediments.

The dominant bedrock types underlying the study basins are listed in Table 2-1.

Methods

In order to assess the impacts of upslope restoration efforts on stream health, aquatic biological and physical data were gathered from 22 sites within the Redwood Creek drainage basin during the spring and summer of 2004 and 2005. These surveys repeated data collection that was conducted in 1973-1975 (Iwatsubo and Averett, 1976). Most of the sites were the same as those sampled by the USGS in the 1970's. Some of the original USGS sites are no longer accessible, so additional sites representing both pristine conditions (Godwood and Upper Prairie Creeks) and basins with recent road removal work (Lost Man Creek and its tributaries) were added to the study base so that a range of conditions was sampled. Sites were all located in Redwood National and State Parks in areas of both old-growth and second-growth coniferous stands. The channels vary from low gradient (< 2 percent) pool-riffle streams to steeper (5 to 9 percent) step-pool streams, and dominant substrate ranges from pebble to boulder.

A 100-m study site was designated at each site near previously established gaging stations. Water discharge, stream gradient, cross-sectional transects, substrate size and amount of large woody debris were measured at all sites in July 2004. Water discharge in tributaries was measured with a pygmy flow meter at each site, and discharge for the mainstem of Redwood Creek was obtained from the USGS gaging station at Orick. The average stream gradient for each study reach was based on longitudinal profiles surveyed with a hand level and stadia rod in steep channels (> 2 percent), or with a self-leveling level in more gentle gradient streams. Cross-sectional transects were surveyed with a stadia rod and measuring tape at previously established sites, if they existed. Where original survey benchmarks could not be located, new cross section survey monuments were established and surveyed. Substrate along the length of each reach was measured using a modified Wolman pebble count technique, by systematic sampling along a tape at one meter spacing (Bunte and Abt, 2001).

To quantify large woody debris, the lengths and widths of all pieces greater than 10 cm in diameter that were within the active channel were recorded. A suspended piece was only counted if it had an influence on the water during peak flows. High flow marks were determined as the tallies were being made. The portion of the wood piece outside of the active channel was not counted. Volume of wood for each piece was calculated as:

$$(\text{length} * 3.14 * (\text{radius}^2))$$

In the case of log jams, where individual pieces could not be measured, the length, width and height of jams were measured. We assumed 50 percent of the log jam volume was wood; the other 50 percent was air space or filled with sediment. Degree of decay, type of wood, and wood zone classifications were not used.

Fine sediment in pools, V*, has been used as an index of sediment supply to the stream channel (Lisle and Hilton 1992). The index is best suited to track changes in fine sediment supply over time within the same stream, rather than comparing values among streams (Hilton and Lisle, 1993), and is only suitable for streams with gradients of 1 to 4 percent and channel substrates of coarse gravel to small cobbles. The average fine sediment load per stream in this study was based on a modification of V* called S*, which is the ratio of maximum sediment depth to maximum residual pool depth (Ashton and others, 2006). Fine sediment in pools was measured in August 2005.

During the past decade Redwood National and State Parks have been monitoring summer temperatures in selected tributaries of Redwood Creek with Hobo temperature recorders. During this study summer water temperatures were monitored at several new sites as well. Data loggers (Onset Computer Corporation, Bourne, Massachusetts, 02532, USA) collected continuous water temperature data in many tributaries during the summers of 1997 – 2005. Data loggers were calibrated prior to use and were deployed from June through September with one-hour sampling intervals. Stream data loggers were submerged in the water column in areas of shade and mixed water. At a subset of stream study sites, we placed air temperature data loggers in a shaded area of the riparian zone in close proximity to the stream data logger. Cool air temperature is a reliable indicator of fog presence during summer months (and consequently less solar radiation is reaching the stream on those days). The accuracy of the probes is ± 0.2°C, and the resolution is ± 0.2°C.

Riparian canopy was measured in 100-m study reaches in the summer of 2005. Percent canopy coverage was estimated at each site where periphyton was measured using a hand-held spherical densiometer. Aerial photos were also compared at all sites between 1978 and 1997 to observe any differences in riparian cover. Photographs and more detailed information on each habitat parameter and stream reach are given in the following sections.

Results

General basin characteristics are listed in Table 2-2.

Table 2-2. General basin and channel characteristics for selected watersheds in Redwood National and State Parks, sorted by percentage of old growth forest.

Sub-Basin	Drainage Area (km^2)	Channel Gradient (%)	Median Particle Size (mm)	Old Growth (%)
Godwood	4.0	1.6	18	100
Hayes	1.5	2.6	58	96
Little Lost Man at gage	8.0	2.6	147	98
Upper Prairie	10.7	1.0	62	94
Little Lost Man at bridge	10.7	0.9	62	89
Harry Weir	7.7	6.5	62	36
Cloquet	2.9	9.3	74	30
Elam	6.7	2.6	30	27
Larry Damm	4.8	0.7	21	21
Lost Man above Larry Damm	24.8	0.8	65	20
Lost Man below North Fork	16.0	--	--	20
Lower Miller	3.4	5.8	58	20
Bond	3.6	6.1	37	18
Bridge	29.0	1.1	57	18
Tom McDonald	17.9	1.1	48	14
South Fork Lost Man	10.2	3.3	72	12
McArthur	9.8	1.0	35	12
North Fork Lost Man	5.8	2.3	53	9
Berry Glen	1.0	4.9	41	7
Fortyfour	8.0	8.0	62	6
Upper Miller	1.8	9.8	49	5
Middle Fork Lost Man	5.9	4.3	52	2

Water Discharge

Many biological and physical measurements are flow dependent, and so the record of water discharge during the sampling periods is important to quantify. The spring and summer flows during 1974 and 2004 were similar (Figure 2-2). Both 1975 and 2005 were higher flow years, with late spring freshets. The spring of 2005 was the wettest of the four years sampled.

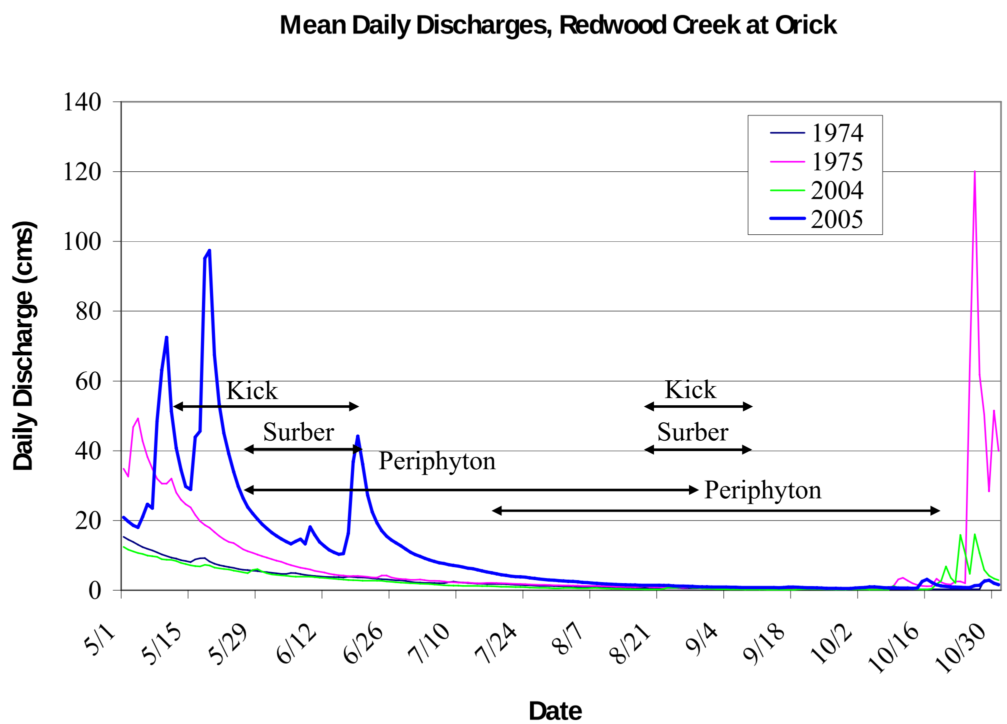


Figure 2-2. Mean daily discharges at Redwood Creek at Orick for the four years of study. Lines with arrows indicate periods of sampling for the biological parameters of interest.

Aquatic biota are also influenced by the magnitude of winter flows preceding the spring sampling season. Figure 2-3 shows that October, 1973 (WY 1974) was wetter than the other years, and April had a large freshet. The highest flow of the sampling period occurred in March, 1975. This peak flow was much higher than in any of the other sampled years. Late April and May of 2005 had higher flows than the other years.

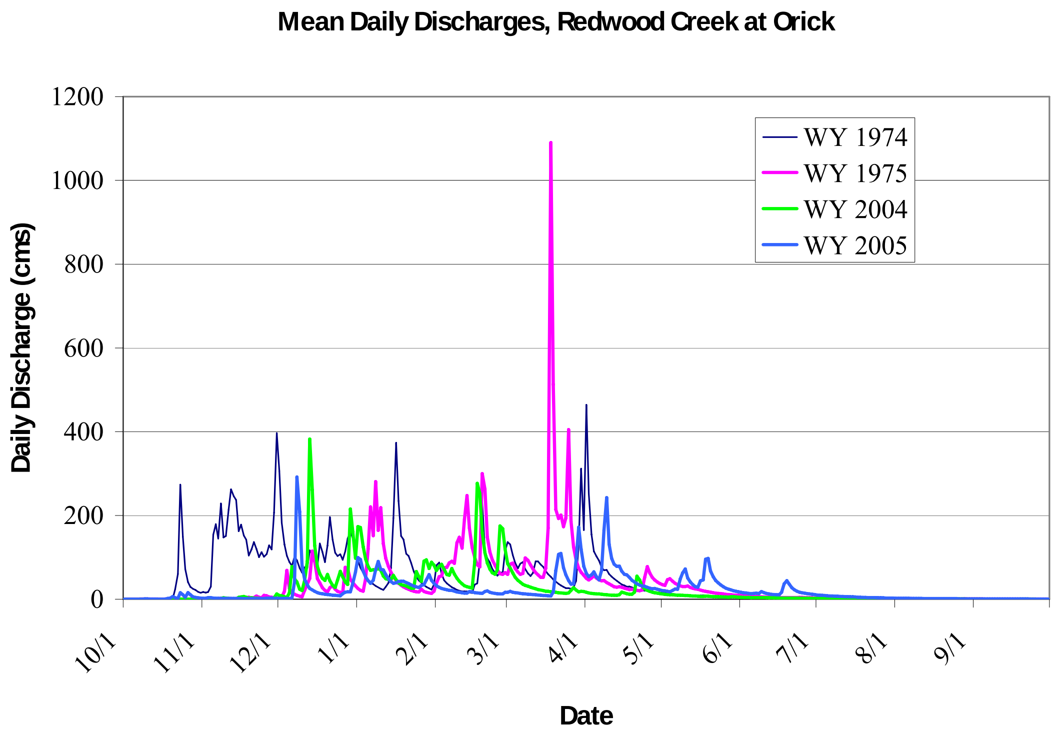


Figure 2-3. Mean daily discharges at Redwood Creek at Orick for the four water years of study.

From a decadal perspective, there were several large flows in the Redwood Creek basin in the 1950's, 1960's and 1970's (Figure 2-4). The last major flow occurred in 1975, just before the 1975 samples were collected. In contrast, the 2004-2005 sampling years followed a series of benign flow years. Redwood National Park's watershed restoration program began in 1978, which coincided with a period of relatively low flows. The 1997 flood was the highest since 1975, but was only a 12-year recurrence interval event. From aerial photographs and channel surveys it is evident that the main channel of Redwood Creek changed more during the floods in the 1970's than in 1997 (Madej, 2001b). It is likely that the tributaries of Redwood Creek were also more influenced by the high floods in the 1970's than the 1997 flood.

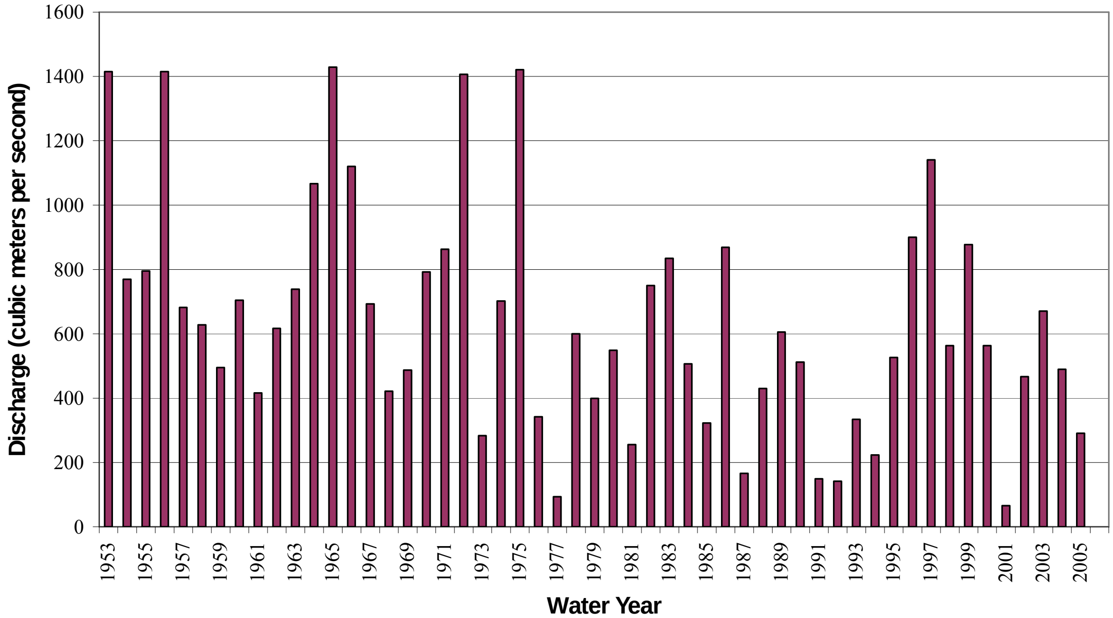


Figure 2-4. Annual peak flows in Redwood Creek at Orick, California.

Water Temperature

Water temperature is an important physical factor that regulates the distribution of fish, amphibians and benthic macroinvertebrates. (Li and others, 1994; Welsh and others, 2001). High water temperatures have been shown to limit the distribution of salmonids within streams (Meisner, 1990), reduce abundance (Ebersole and others, 2001) and fragment populations within a watershed (Matthews and Zimmerman, 1990). Elevated water temperatures can also decrease growth and increase juvenile mortality (Brett, 1979). High water temperatures can negatively influence salmonid egg development, juvenile appetite and growth (Dockray and others, 1996), as well as negatively alter behavior and inter-species interactions (De Staso and Rahel 1994; Beschta and others, 1987). For the nearby Mattole River basin Welsh and others (2001) reported that juvenile coho were not present in streams where the maximum weekly maximum temperature (MWMT) exceeded 18.1°C or where the maximum weekly average temperature (MWAT) exceeded 16.8°C.

Table 2-3 reports temperature maxima for the years of record. MWAT was originally defined by Brungs and Jones (1977) as a limit to the maximum temperature for a species and life stage that should not be exceeded in a regulatory sense, but more recently is the term used to describe the maximum value of a seven-day moving average of daily average temperatures (Welsh and others, 2001). This latter usage is the definition used in the present study. Similarly, MWMT is the maximum value of a seven-day moving average of daily maximum temperatures.

Table 2-3. Summary of water temperature data from July 1 to August 31 for years of record in tributaries of Redwood Creek.

Stream	Area (km^2)	Year	Maximum Temp (°C)	MWAT (°C)[a]	MWMT (°C)[b]	Maximum Diurnal Range (°C)
Bridge	29.0	1997	19.3	16.3	18.9	4.9
		1998[c]	18.3	16.0	17.8	4.1
		1999[d]	18.5	16.1	18.0	4.5
		2000	18.2	16.0	17.9	3.8
		2001	18.2	15.8	17.6	3.9
		2002	17.2	15.5	16.9	3.6
		2003	17.7	16.0	17.4	3.3
		2004	19.0	16.6	18.5	3.5
		2005	17.0	15.8	16.9	2.7
Prairie at Wolf Creek	32.6	1998	17.4	16.0	16.8	3.1
		1999	16.1	15.3	15.7	3.3
		2000	16.4	15.3	16.0	3.0
		2001	16.1	15.1	15.8	2.5
		2002	16.3	15.1	15.8	3.1
		2003	16.6	15.8	16.3	2.7
		2004	16.8	16.0	16.6	2.4
		2005	16.6	15.4	16.4	2.8
Lost Man at Hatchery	32.0	2000	17.0	15.0	16.1	3.3
		2001	16.4	14.8	15.9	3.9
		2002	16.4	14.4	15.6	3.8
		2003	16.9	15.5	16.6	3.3
		2004	17.0	15.7	16.8	3.1
		2005	16.7	15.2	16.3	3.0
Larry Damm	4.8	2000	14.6	13.9	14.2	1.7
		2001	14.3	13.6	13.8	1.6
		2002	14.0	13.3	13.5	1.7
Tom McDonald	17.9	2001[e]	15.7	14.6	15.3	2.4
		2002	15.5	14.3	15.1	2.6
		2003	16.0	14.6	15.7	2.8
		2004	16.3	15.1	16.1	2.5
		2005	16.1	14.7	16.0	2.4

Table 2-3. continued.

Stream	Area (km^2)	Year	Maximum Temp (°C)	MWAT (°C)	MWMT (°C)	Maximum Diurnal Range (°C)
Little Lost Man at gage	8.0	2003	17.0	15.4	16.7	3.5
		2004	17.3	15.9	17.2	3.5
		2005	16.8	14.9	16.3	2.8
Bond	3.6	2004	14.3	14.0	14.0	0.8
Cloquet	2.9	2004	15.4	15.0	15.3	1.3
Upper Miller	1.8	2005	15.0	14.6	14.8	1.6
North Fork Lost Man	5.8	2005	14.9	14.0	14.3	1.6
Middle Fork Lost Man	5.9	2005	14.8	14.3	14.3	1.3
South Fork Lost Man	10.2	2005	14.3	13.7	14.2	1.5
Harry Weir	7.7	2005	16.2	15.4	15.9	1.6
Fortyfour	8.0	2005	14.5	13.9	14.3	1.4

a = MWAT is the maximum weekly average temperature measured from July 1 to August 31.
b = MWMT is the maximum weekly maximum temperature measured from July 1 to August 31.
c = temperature only recorded from 7/29/98-8/31/98
d = temperature only recorded from 7/14/99-8/31-99
c = temperature only recorded from 7/26/01-8/31-01

Water temperatures reached their summer maxima in late July or early August for all years of record. No tributaries exceeded an MWAT value of 16.8°C, the level of concern for juvenile coho. For the period of record, maximum weekly average temperatures (MWAT) and maximum weekly maximum temperatures (MWMT) were highest at Bridge Creek, which has the largest drainage area and is farthest inland of all the study sites. Bridge Creek was the only tributary that exceeded an MWMT value of 18.1°C (in 1997 and in 2004). In contrast, in the mainstem of Redwood Creek in Redwood Valley, MWATs and MWMTs for several years were >21.8°C and > 25°C, respectively (Madej and others, in press). Studies by Welsh and others (2001) on tributaries of the Mattole River, California suggest that streams with MWMT greater than 18.1°C or MWAT greater than 16.8°C may restrict the presence of juvenile coho salmon. (Coho salmon in Redwood Creek and the Mattole River are grouped within the same Evolutionary Significant Unit). Consequently, the tributaries measured in this study represent possible cool water refugia for juvenile coho.

Stream temperature controls the developmental rate of aquatic organisms. Invertebrates require a certain amount of heat to develop from one point in their life cycles to another, and such physiological time is commonly expressed in degree-day units. Unfortunately, daily stream temperature data are not available for the sampled streams from 1974 and 1975, so a comparison of recent thermal regimes with that of the 1970’s is not available.

Cross-sectional and Longitudinal Profile Surveys

In 1974 the USGS established cross-sectional transects on the major tributaries of Redwood Creek. They were re-surveyed annually for many years, but most cross sections were not surveyed since the late 1980’s. In 2004 we tried to relocate the cross section monuments and resurvey these sites. If the survey endpoints could not be found, earlier photographs were compared with current channel conditions. In most tributaries, most of the deposits from the 1975 flood had been flushed out by 1986 (Madej, 1987). The 2004 surveys showed little change since the 1986 surveys. In general, tributary channel recovery corresponded with channel gradient, and steeper channels had less stored sediment. New cross-sectional transects established in the current study will allow quantification of future channel aggradation or degradation in the tributary channels.

Longitudinal profiles were surveyed for each study reach in 2004 in order to calculate stream gradient, and these profiles can be used as a baseline for future monitoring. The only stream with an earlier thalweg profile was Bridge Creek. In that reach, the percent of channel length in pools increased from 1986 to 1997, and maximum pool depths also increased (Madej, 1999).

Sediment Loading

Sediment affects aquatic biota in many ways. Fine sediment can fill the interstices of a gravel-bedded channel, leading to decreased oxygenation of the gravels, or can impede fry emergence. Fine sediment can cover suitable spawning gravel. Sediment can fill pools, decreasing the amount of rearing and hiding habitat for salmonids. Large amounts of fine sediment reduce or eliminate much of the suitable substrate for producing macroinvertebrates, thereby limiting the food available to juvenile fish (Furniss and others, 1991).

Although the USGS collected suspended sediment samples in the mainstem of Redwood Creek basin in the 1970's, very few data exist for the tributaries of Redwood Creek that were the focus of this current study. We did analyze sediment trends detected at the Redwood Creek at Orick gaging station, however. Sediment rating curves express the rates of suspended sediment transport as a function of flow magnitude. A downward shift in the intercept of the regression lines of this relationship represents a lower sediment concentration for a given discharge, whereas a shift in the slope represents a change in the rate of sediment transport with increasing discharge. A comparison of suspended sediment-discharge rating curves shows a significant shift from higher sediment transport rates in the 1970's to lower rates in the 1980's and the 1990's, and even lower sediment transport rates in the 2000's (Appendix 2-2). A more detailed account of sediment transport trends is found in Madej and others (2006). We assume that the decreased sediment loading in the mainstem of Redwood Creek is also true for many of the tributaries of Redwood Creek in this study, in which timber harvest has ceased and abandoned roads have been removed. Newly established gaging stations on several streams in Redwood National and State Parks will help clarify tributary sediment loading in the near future.

The amount of fine sediment in pools, indexed by V* and its modified version S*, is listed under "General Channel Conditions" (below). Many of the streams in this study were too steep or coarse to be able to use V* or S* as an index. Streams in which we measured fines were generally in the range of 0.3 to 0.5. Larry Damm Creek had the highest S* ratio (0.9). This stream drains the sandy Gold Bluffs unit of the Prairie Creek Formation, and the basin has also had the most intensive recent road removal work (Figure 1-4).

Turbidity has been measured in sub-basins of Lost Man Creek with and without recent road removal work. Klein (2006) concluded that 1) turbidity often (but not always) increased as water flowed through stream crossing excavations, 2) downstream (offsite) peak storm turbidities were often higher in tributaries draining areas with recent road removal than in tributaries with little or no recent disturbance, 3) erosion at new stream crossing excavations was generally most severe in the earliest large storms of the first season and diminished as the winter runoff season progressed. That study also showed that Larry Damm Creek was a disproportionately large source of suspended sediment and turbidity to downstream areas in Water years 2003 and 2004, but its relative contribution had diminished considerably in 2005, suggesting that high erosion rates from road decommissioning work was short lived. To date, no biological monitoring of the excavated stream crossings sites has been undertaken, so the impacts of elevated turbidity on aquatic biota are currently unknown.

Canopy Closure

The amount of riparian canopy is an important factor regulating temperature, light, organic matter and nutrient inputs into a stream and affects the processes of all living aquatic organisms. It is one of many important factors determining periphyton growth. Riparian stand composition varied from alder dominated to redwood dominated, but the distinction was not always clear. For example, Little Lost Man Creek at the gage is located in an uncut redwood forest, but the riparian zone within 15 m of the streambank is dominated by alders and ferns.

General Channel Conditions

The following photographs and tables depict channel conditions at the sampling sites in the summer of 2004, summarize physical habitat conditions, and illustrate changes in riparian condition from air photo comparisons. The scale of the air photos is about 1:6000.

Figure 2-5a. Bridge Creek channel characteristics 7/8/04

Substrate Size:

D_{84}: 237 mm

D_{50}: 57 mm

D_{16}: 6 mm

< 4mm: 8%

Number of Wood Pieces /100m: 36

Number of Log Jams/100m: 0

Volume of In-Channel Wood: 146m^3/100m

Discharge: 0.1 cms

Riparian Cover: 85%

S*: 0.3

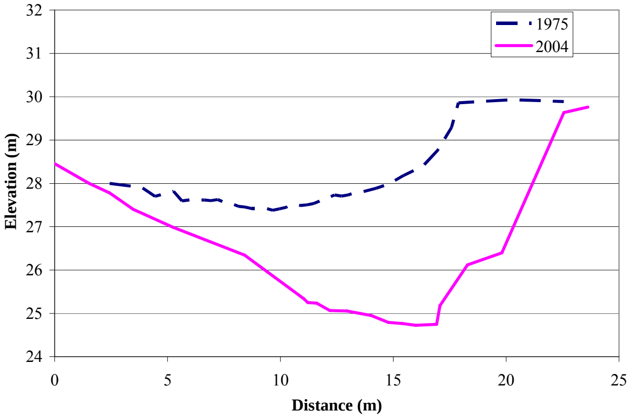


Fig. 2-5b. Larry Damm Creek channel characteristics 7/9/04

Substrate Size:

D_{84}: 71 mm

D_{50}: 21 mm

D_{16}: 6 mm

< 4mm: 13%

Number of Wood Pieces /100m: 43

Number of Log Jams/100m: 0

Volume of In-Channel Wood: 20 m^3/100m

Discharge: 0.02 cms

Riparian Cover: 98%

S*: 0.9

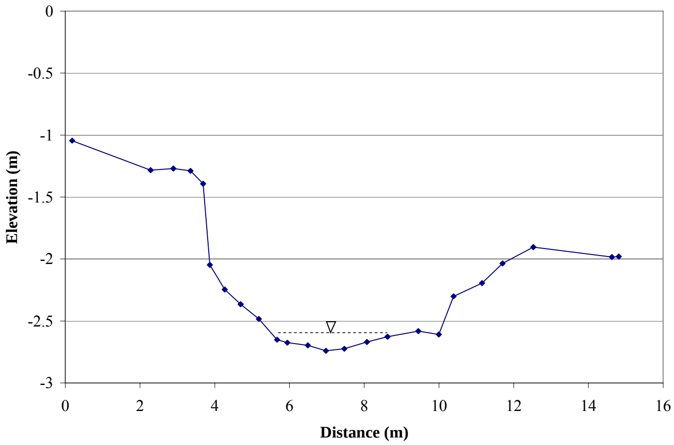


Fig. 2-5c. Hayes Creek channel characteristics 7/12/04

Substrate Size:

D_{84}: 159 mm

D_{50}: 58 mm

D_{16}: 18 mm

< 4mm: 9%

Number of Wood Pieces /100m: 12

Number of Log Jams/100m: 1

Volume of In-Channel Wood:
22 m^3/100m

Discharge: na

Riparian Cover: 98%

S*: 0.3

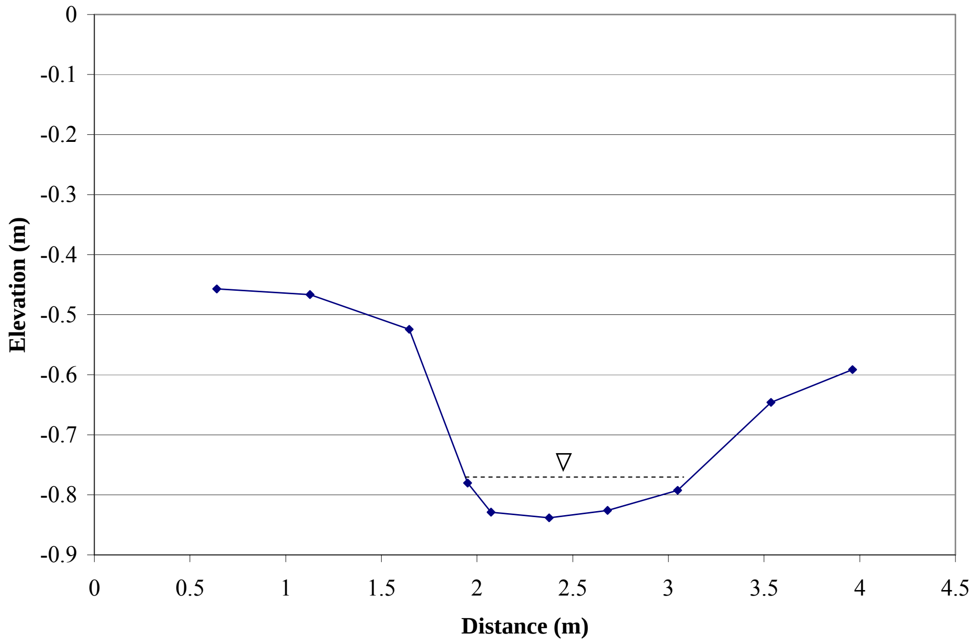


Fig 2-5d. McArthur Creek channel characteristics 7/12/04

Substrate Size:

D_{84}: 69 mm

D_{50}: 35 mm

D_{16}: 14 mm

< 4mm: 7%

Number of Wood Pieces /100m: 14

Number of Log Jams/100m: 0

Volume of In-Channel Wood: 12 m^3/100m

Discharge: 0.04 cms

Riparian Cover: 94%

S*: 0.8

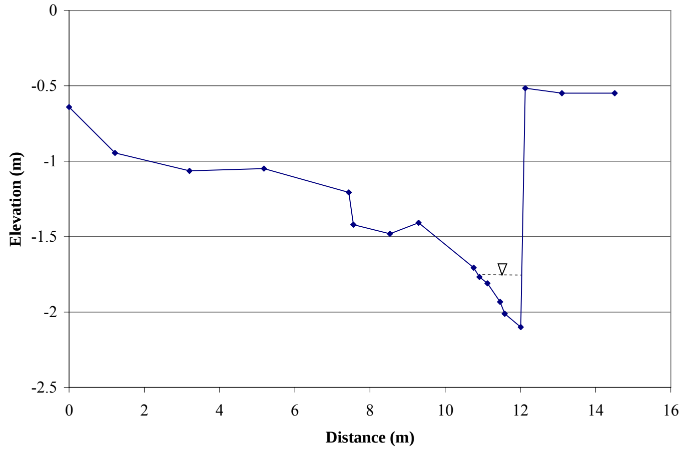


Fig 2-5e. Elam Creek channel characteristics 7/13/04

Substrate Size:

D_{84}: 54 mm

D_{50}: 30 mm

D_{16}: 6 mm

< 4mm: 14%

Number of Wood Pieces /100m: 36

Number of Log Jams/100m: 3

Volume of In-Channel Wood: $165m^3$ /100m

Discharge: 0.04 cms

Riparian Cover: 96%

S*: 0.6

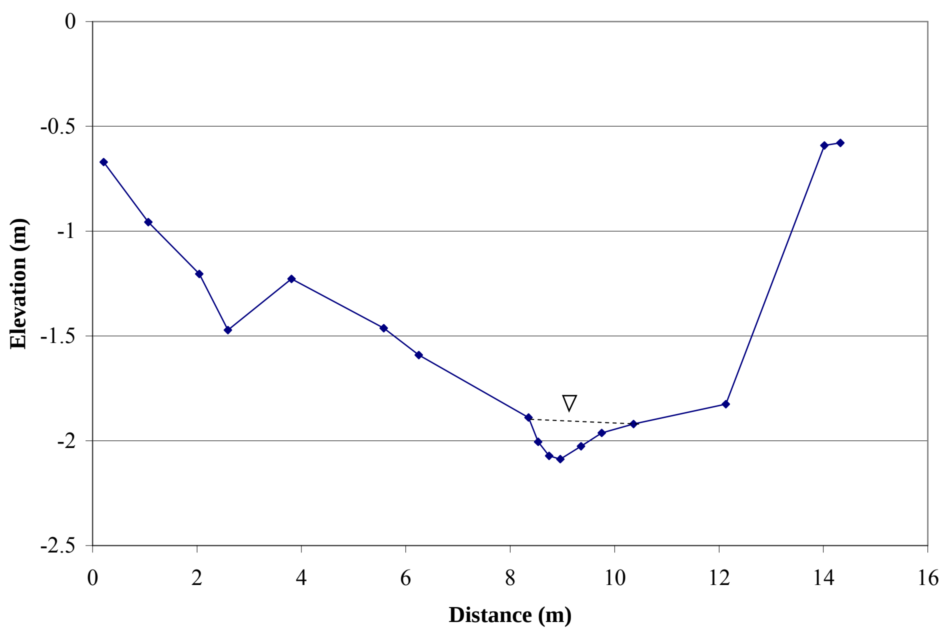


Fig 2-5f. Harry Weir Creek channel characteristics 7/14/04

Substrate Size:

D_{84}: 237 mm

D_{50}: 62 mm

D_{16}: 16 mm

< 4mm: 6%

Number of Wood Pieces /100m: 32

Number of Log Jams/100m: 4

Volume of In-Channel Wood: 141 m^3/100m

Discharge: 0.02 cms

Riparian Cover: 96%

S*: na

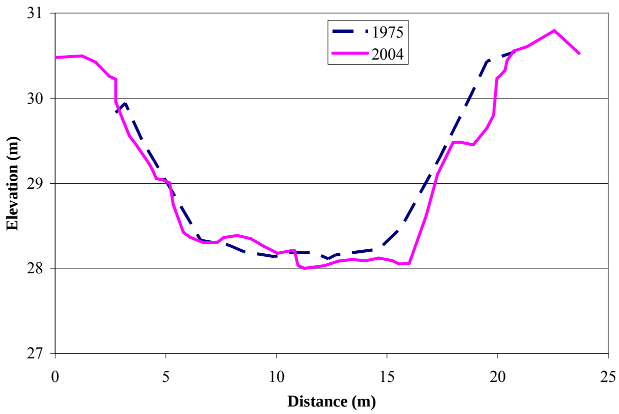


Fig. 2-5g. Bond Creek channel characteristics 7/20/04

Substrate Size:

D_{84}: 83 mm

D_{50}: 37 mm

D_{16}: 12 mm

< 4mm: 9%

Number of Wood Pieces /100m: 27

Number of Log Jams/100m: 1

Volume of In-Channel Wood: 71 m^3/100m

Discharge: 0.02 cms

S*: na

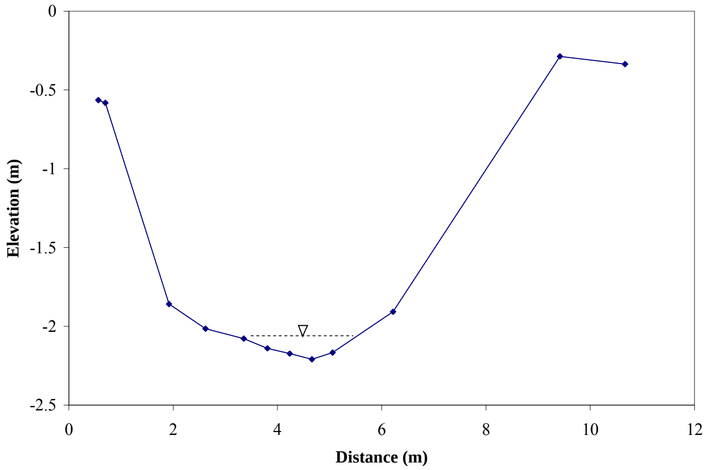


Fig. 2-5h. Upper Miller Creek channel characteristics 7/28/04

Substrate Size:

D_{84}: 199 mm

D_{50}: 49 mm

D_{16}: 13 mm

< 4mm: 9%

Number of Wood Pieces /100m: 15

Number of Log Jams/100m: 4

Volume of In-Channel Wood: 335 m^3/100m

Discharge: 0.01 cms

Riparian Cover: 97%

S*: na

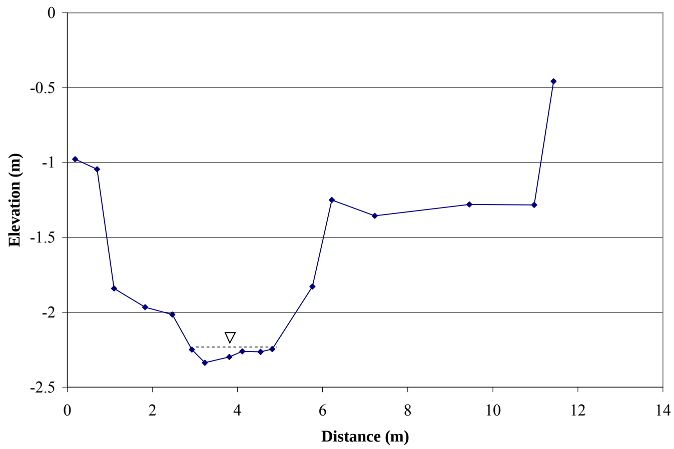


Fig. 2-5i. Lower Miller Creek channel characteristics 7/20/04

Substrate Size:

D_{84}: 171 mm

D_{50}: 58 mm

D_{16}: 23 mm

< 4mm: 2%

Number of Wood Pieces /100m: 23

Number of Log Jams/100m: 1

Volume of In-Channel Wood: 66m^3/100m

Discharge: 0.00 cms

Riparian Cover: 97%

S*: na

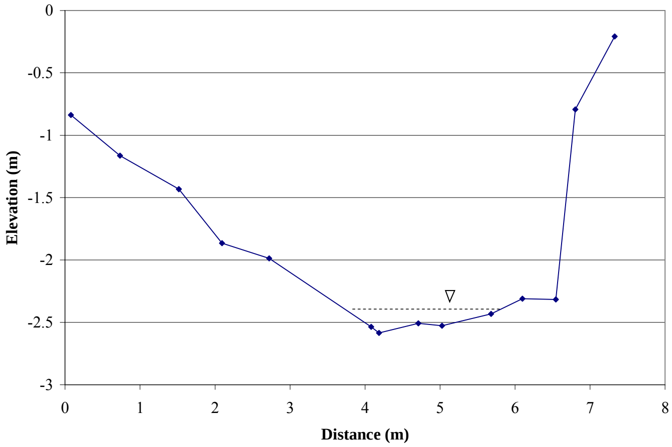


Fig. 2-5j. Berry Glen Creek channel characteristics 7/27/04

Substrate Size:

D_{84}: 137 mm

D_{50}: 41 mm

D_{16}: 13 mm

< 4mm: 8%

Number of Wood Pieces /100m: 19

Number of Log Jams/100m: 0

Volume of In-Channel Wood: 4 m^3/100m

Discharge: na

Riparian Cover: 98%

S*: 0.7

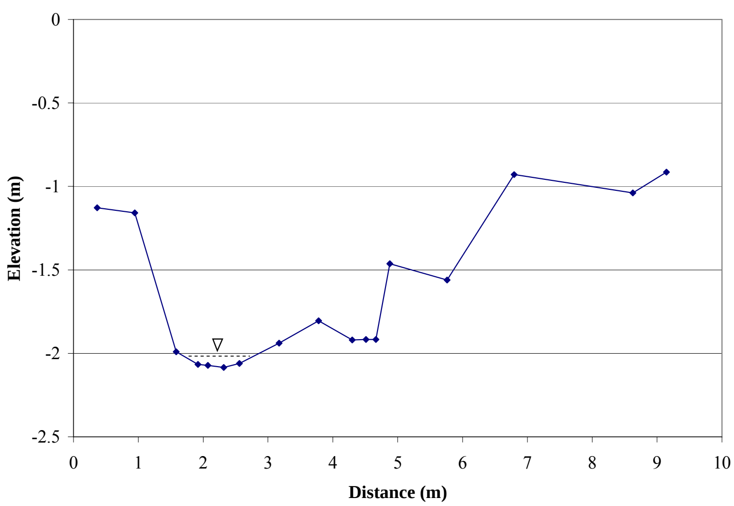


Fig. 2-5k. North Fork Lost Man Creek channel characteristics 7/27/04

Substrate Size:

D_{84}: 205 mm

D_{50}: 53 mm

D_{16}: 14 mm

< 4mm: 13%

Number of Wood Pieces /100m: 20

Number of Log Jams/100m: 0

Volume of In-Channel Wood: 18 m^3/100m

Discharge: 0.01 cms

Riparian Cover: 97%

S*: 0.4

Fig. 2-5l. South Fork Lost Man Creek channel characteristics 7/22/04

Substrate Size:

D_{84}: 343 mm

D_{50}: 72 mm

D_{16}: 16 mm

< 4mm: 6%

Number of Wood Pieces /100m: 41

Number of Log Jams/100m: 0

Volume of In-Channel Wood: 98 m^3/100m

Discharge: 0.02 cms

S*: 0.3

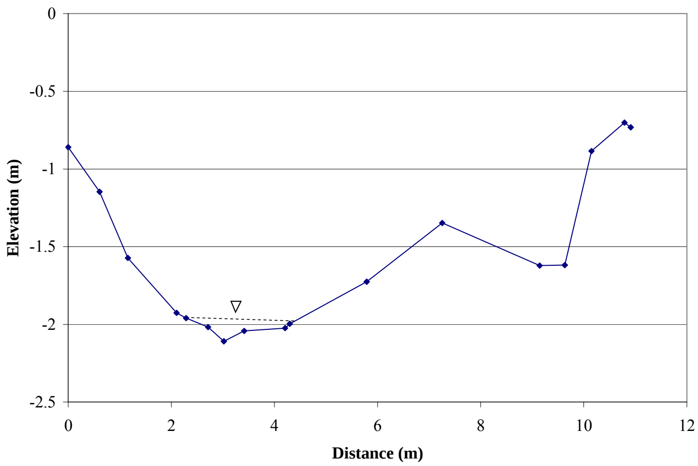
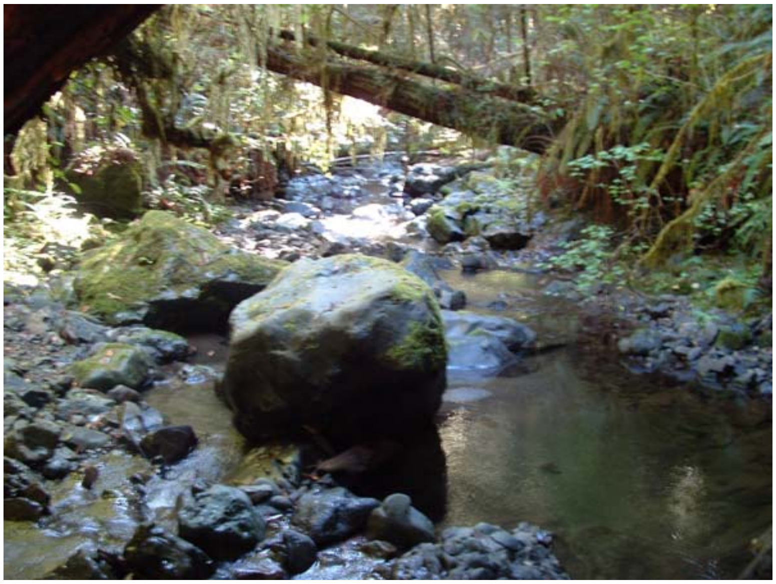


Fig. 2-5m. Middle Fork Lost Man Creek channel characteristics 7/22/04



Substrate Size:

D_{84}: 261 mm

D_{50}: 52 mm

D_{16}: 11 mm

< 4mm: 9%

Number of Wood Pieces /100m: 38

Number of Log Jams/100m: 0

Volume of In-Channel Wood: 106 m^3/100m

Discharge: 0.03 cms

S*: 0.4

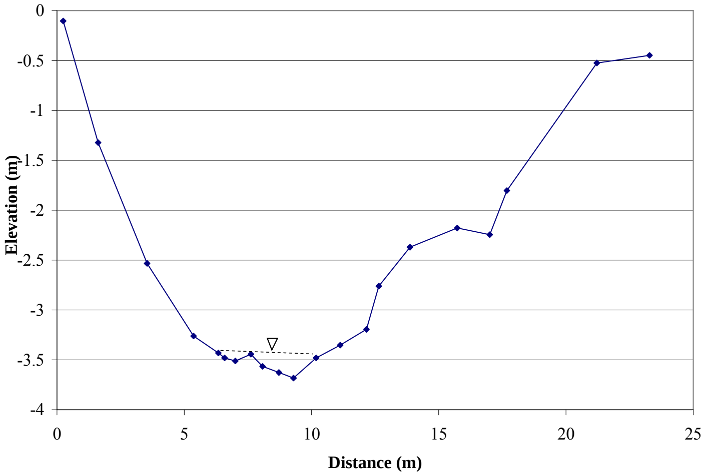


Fig. 2-5n. Godwood Creek channel characteristics

Substrate Size:

D_{84}: 40

D_{50}: 18

D_{16}: 6

Number of Wood Pieces /100m: 25

Number of Log Jams/100m: 0

Volume of In-Channel Wood: 56 m^3/100m

S*: 0.5

Fig 2-5o. Lost Man Creek upstream of Larry Damm Creek channel characteristics 7/19/04

Substrate Size

D_{84}: 101mm

D_{50}: 65mm

D_{16}: 32mm

< 4mm: 2%

Number of Wood Pieces /100m: 17

Number of Log Jams/100m: 0

Volume of In-Channel Wood: 59m^3/100m

Discharge: na

Riparian Cover: 94%

S*: 0.3

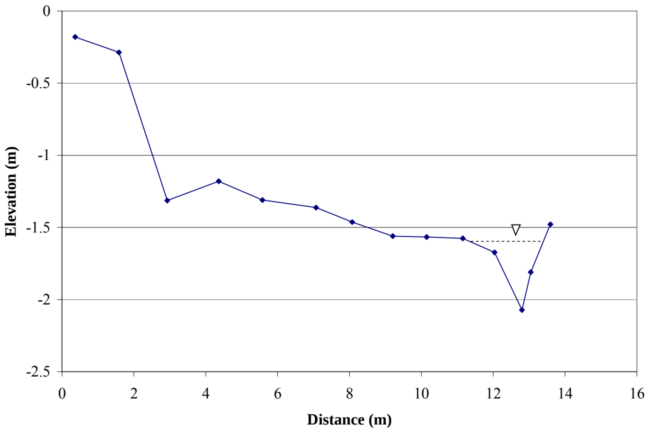


Fig. 2-5p. Lost Man Creek downstream of North Fork Lost Man Creek channel characteristics 7/23/04

Substrate Size:

D_{84}: 335mm

D_{50}: 119mm

D_{16}: 33mm

< 4mm: 3%

Number of Wood Pieces /100m: 4

Number of Log Jams/100m: 0

Volume of In-Channel Wood: $21m^3/100m$

Discharge: 0.06 cms

Riparian Cover: 67%

S*: 0.3

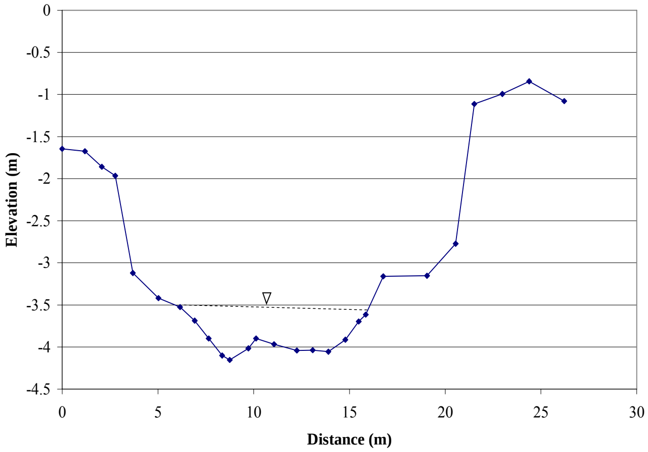


Fig. 2-5q. Little Lost Man Creek at the bridge channel characteristics 7/21/04

Substrate Size:
D_{84}: 180mm
D_{50}: 62mm
D_{16}: 25mm
< 4mm: 3%
Number of Wood Pieces /100m:
3
Number of Log Jams/100m: 0
Volume of In-Channel Wood: 3 m^3/100m
Discharge: 0.01 cms
Riparian Cover: 97%
S*: 0.2

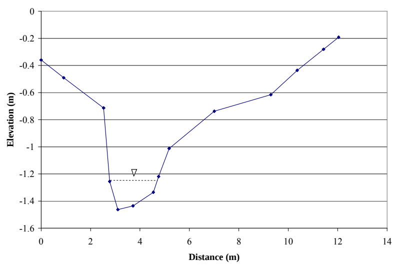


Fig. 2-5r. Little Lost Man Creek at the gage channel characteristics 7/21/04

Substrate Size:

D_{84}: 406mm

D_{50}: 147mm

D_{16}: 29mm

< 4mm: 3%

Number of Wood Pieces /100m: 10

Number of Log Jams/100m: 1

Volume of In-Channel Wood: 215 m^3/100m

Discharge: 0.04 cms

Riparian Cover: 98%

S*: 0.4

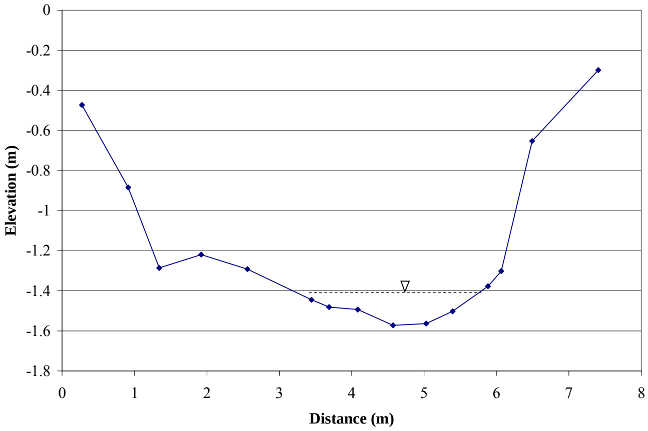


Fig. 2-5s. Fortyfour Creek channel characteristics 7/15/04

Substrate Size:

D_{84}: 417mm

D_{50}: 62mm

D_{16}: na

< 4mm: 23%

Number of Wood Pieces /100m: 48

Number of Log Jams/100m: 1

Volume of In-Channel Wood: 18 m^3/100m

Discharge: 0.05 cms

Riparian Cover: 99%

S*: na

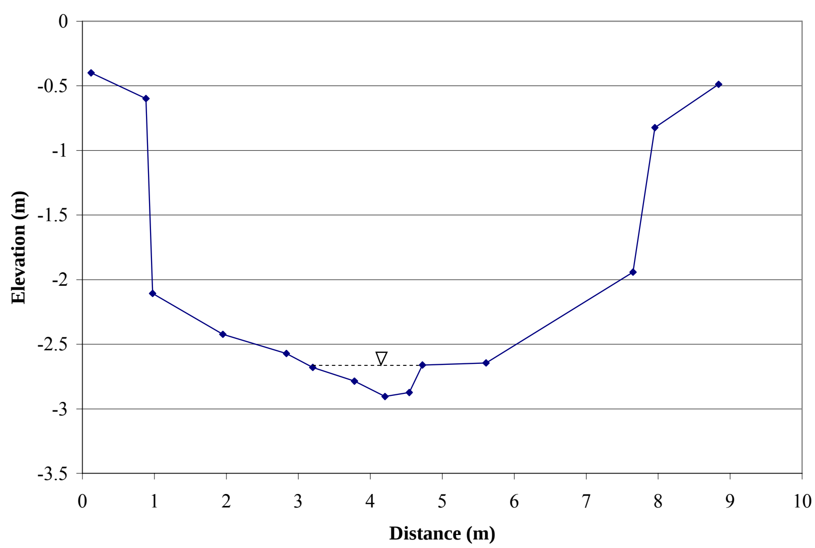


Fig 2-5t. Upper Prairie Creek channel characteristics 7/23/04

Substrate Size:

D_{84}: 135mm

D_{50}: 62mm

D_{16}: 30mm

< 4mm: 1%

Number of Wood Pieces /100m: 8

Number of Log Jams/100m: 0

Volume of In-Channel Wood: 1 m^3/100m

Discharge: 0.05 cms

S*: 0.2

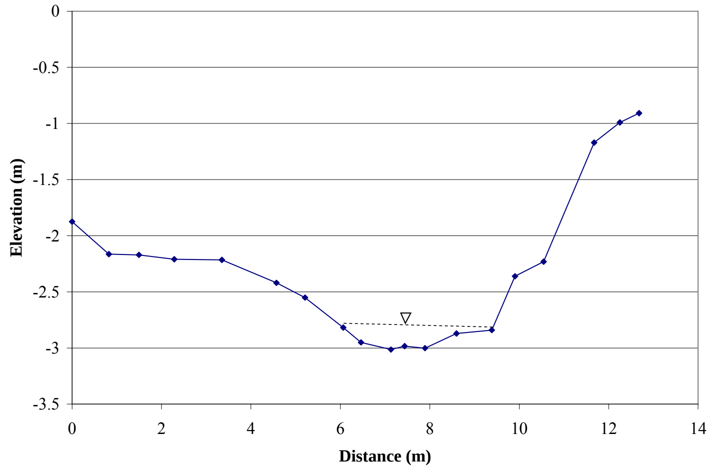


Fig. 2-5u. Cloquet Creek channel characteristics 7/21/04

Substrate Size:

D_{84}: 271mm

D_{50}: 74mm

D_{16}: 16mm

< 4mm: 8%

Number of Wood Pieces /100m: 55

Number of Log Jams/100m: 0

Volume of In-Channel Wood: 61m^3/100m

Discharge: 0.001 cms

S*: na

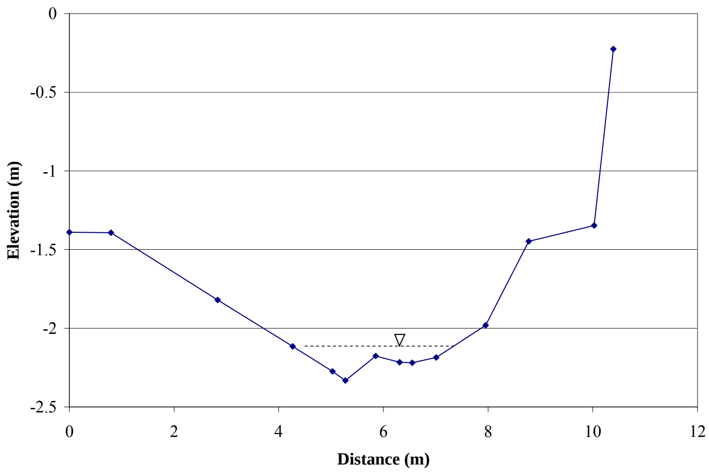


Fig. 2-5v. Tom McDonald Creek channel characteristics 7/7/04

Substrate Size:

D_{84}:150mm

D_{50}: 48mm

D_{16}: 14mm

< 4mm: 4%

Number of Wood Pieces /100m: 18

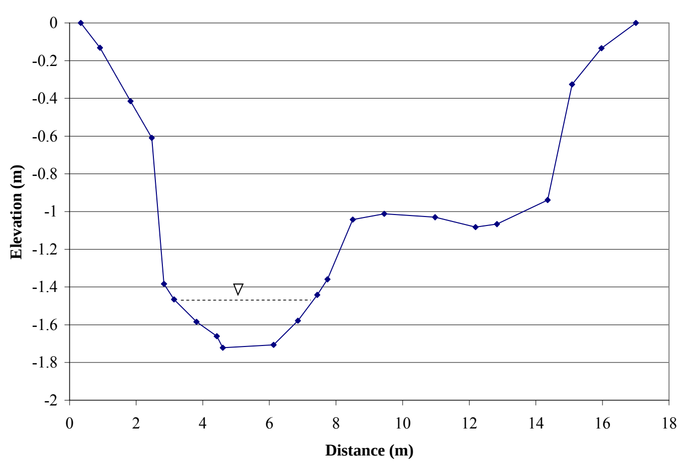
Number of Log Jams/100m: 0

Volume of In-Channel Wood: 11 m^3/100m

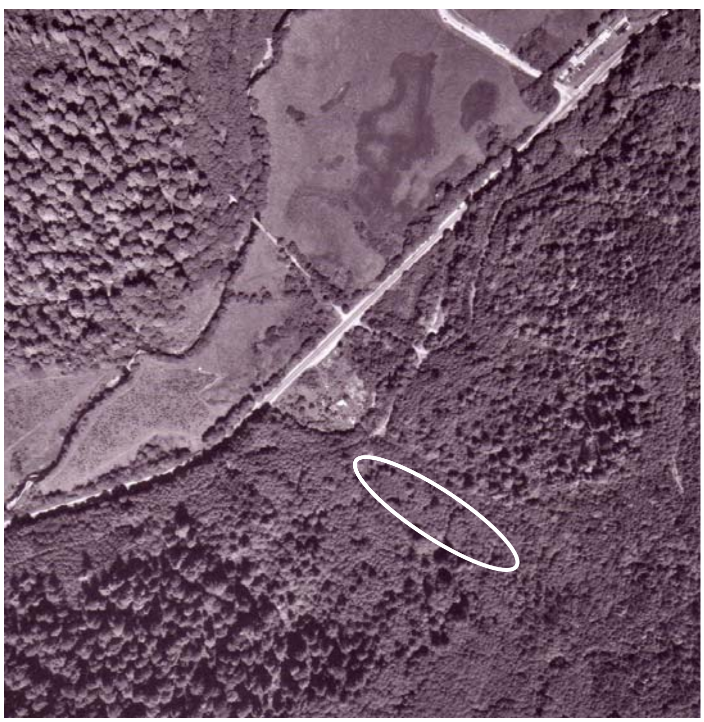
Discharge: 0.07 cms

Riparian Cover: 88%

S*: 0.7



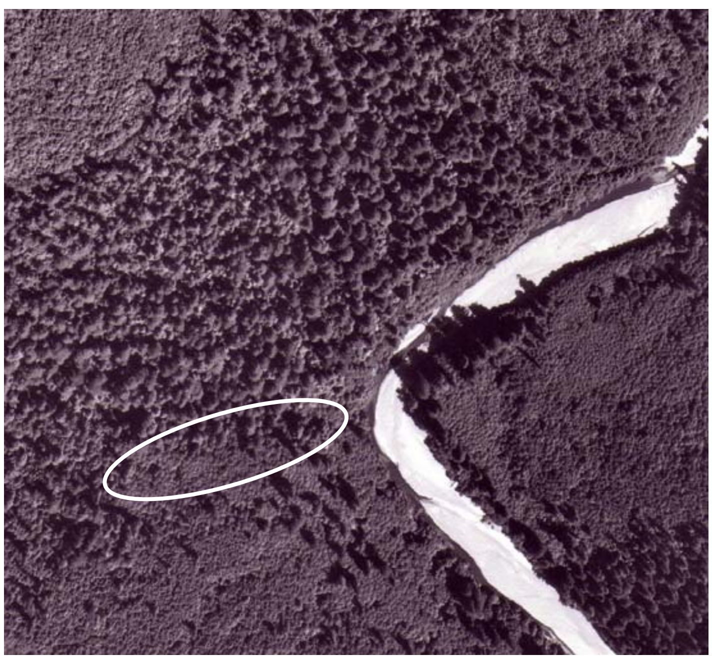
1978



1997

Figure 2-6a. Air photo comparisons of Berry Glen Creek in 1978 and 1997.

1978



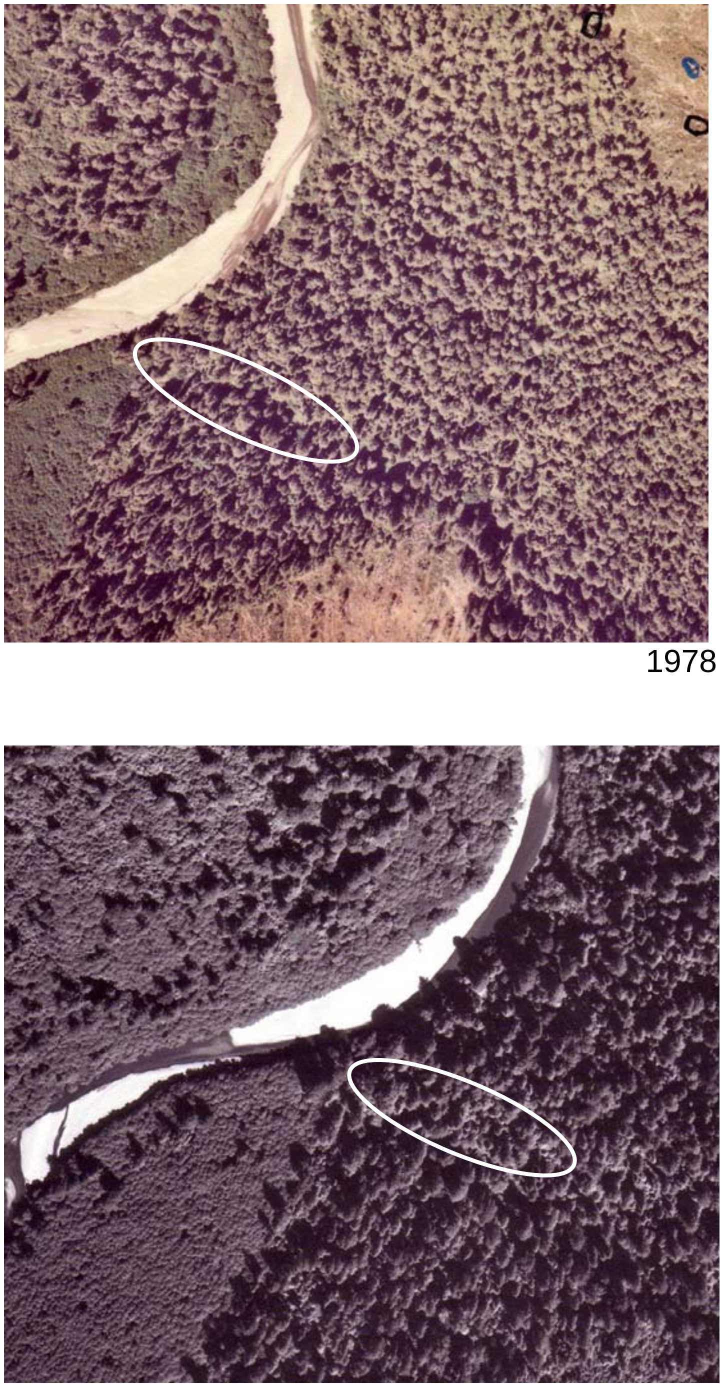
1997

Figure 2-6b. Air photo comparisons of Bond Creek in 1978 and 1997.

1978

1997

Figure 2-6c. Air photo comparisons of Bridge Creek in 1978 and 1997.

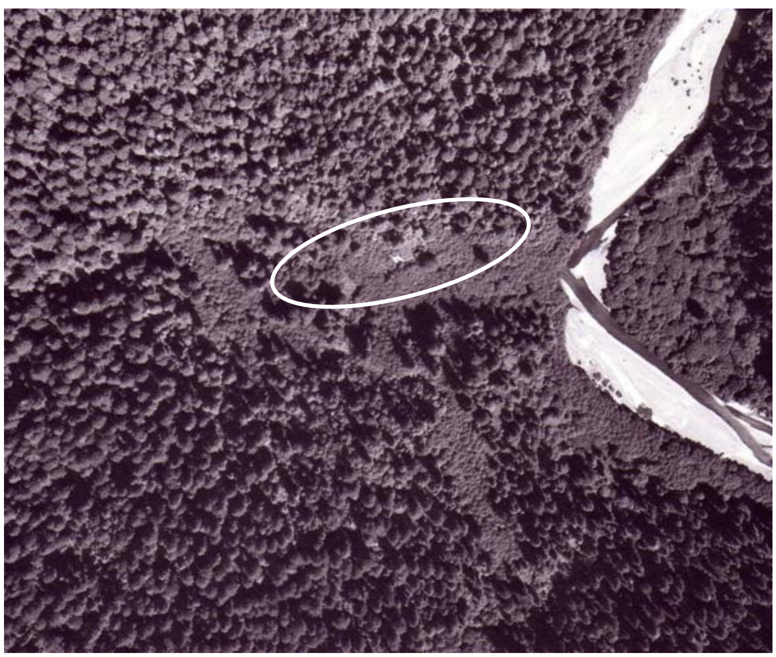


1978

1997

Figure 2-6d. Air photo comparisons of Cloquet Creek in 1978 and 1997.

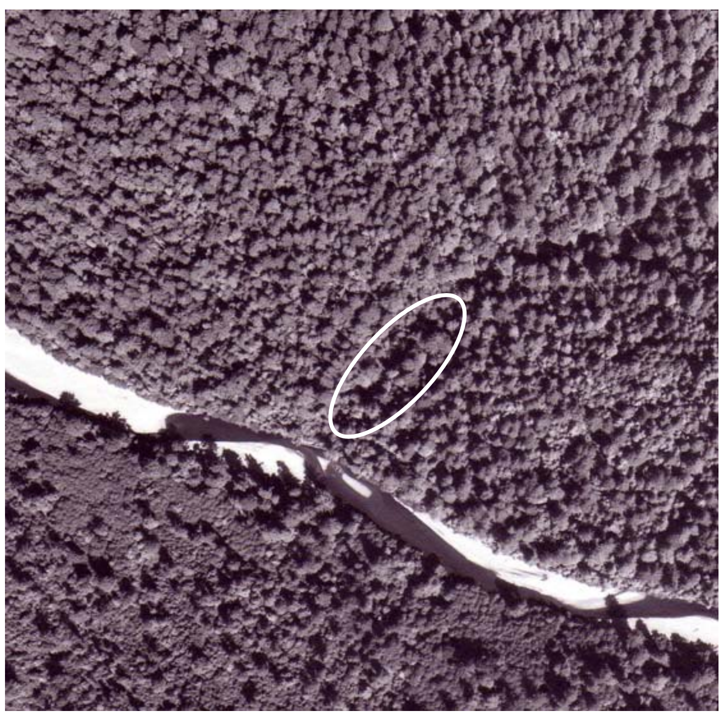
1978



1997

Figure 2-6e. Air photo comparisons of Elam Creek in 1978 and 1997.

1978



1997

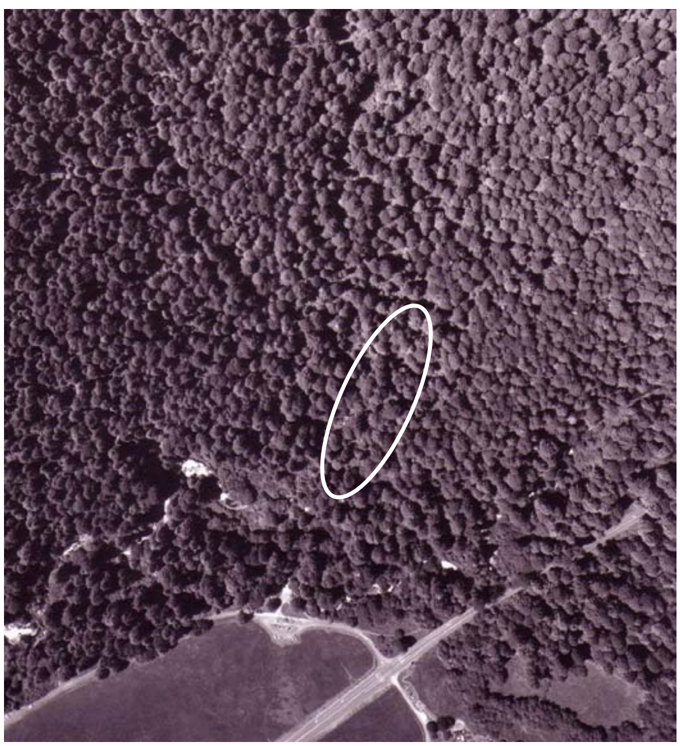
Figure 2-6f. Air photo comparisons of Harry Weir Creek in 1978 and 1997.

1978

1997

Figure 2-6g. Air photo comparisons of Fortyfour Creek in 1978 and 1997.

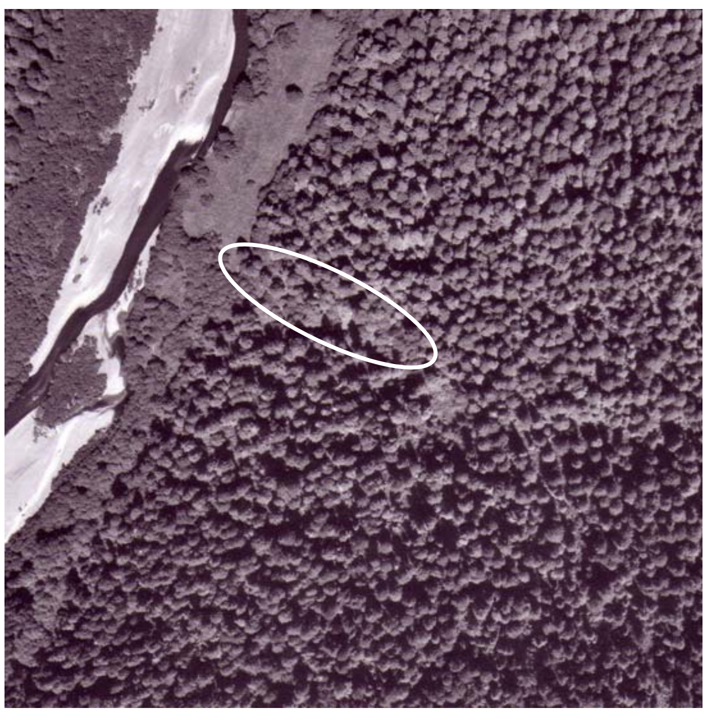
1978



1997

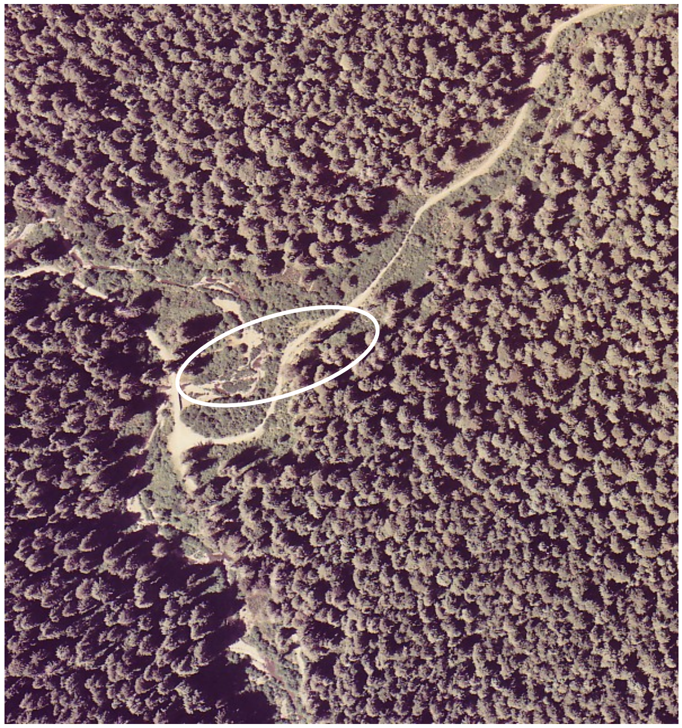
Figure 2-6h. Air photo comparisons of Godwood Creek in 1978 and 1997.

1978



1997

Figure 2-6i. Air photo comparisons of Hayes Creek in 1978 and 1997.



1978

1997

Figure 2-6j. Air photo comparisons of Lost Man Creek upstream of Larry Damm Creek in 1978 and 1997.

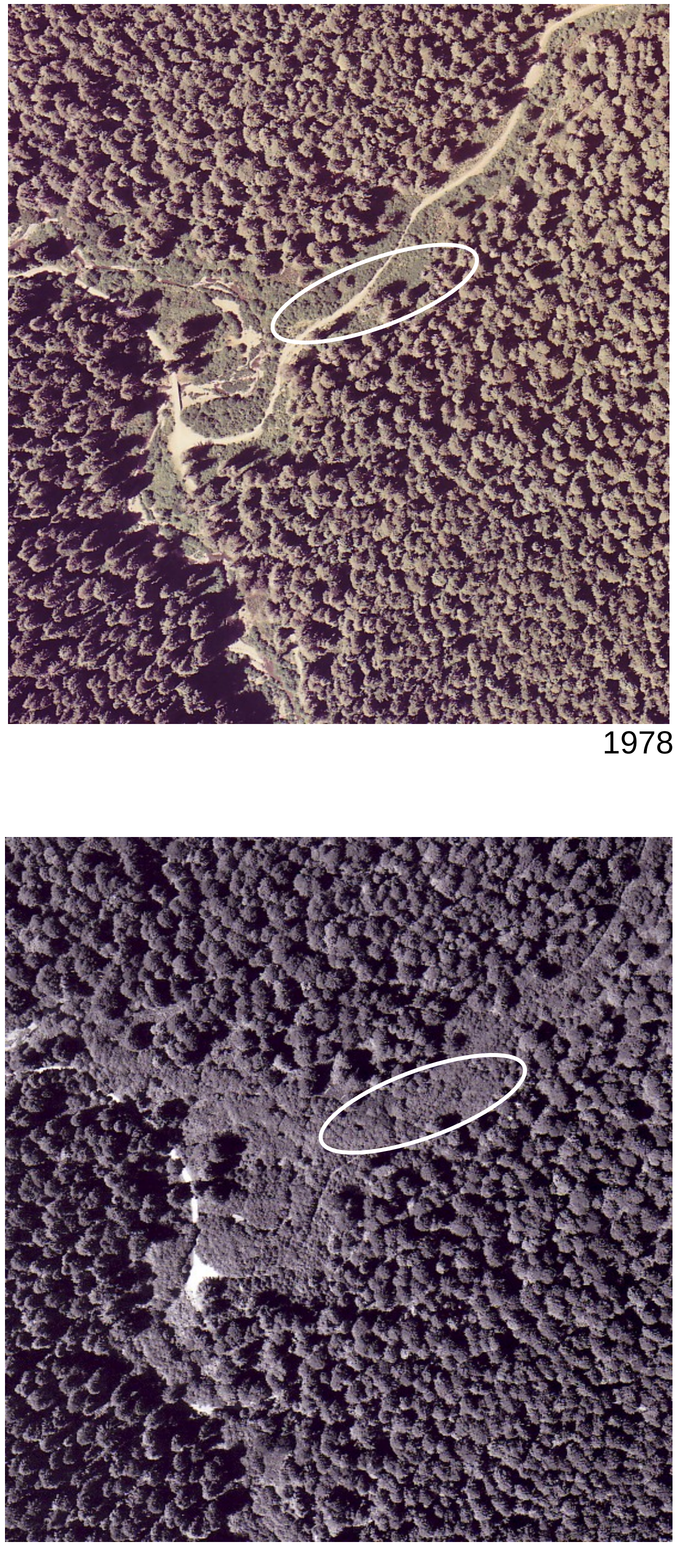
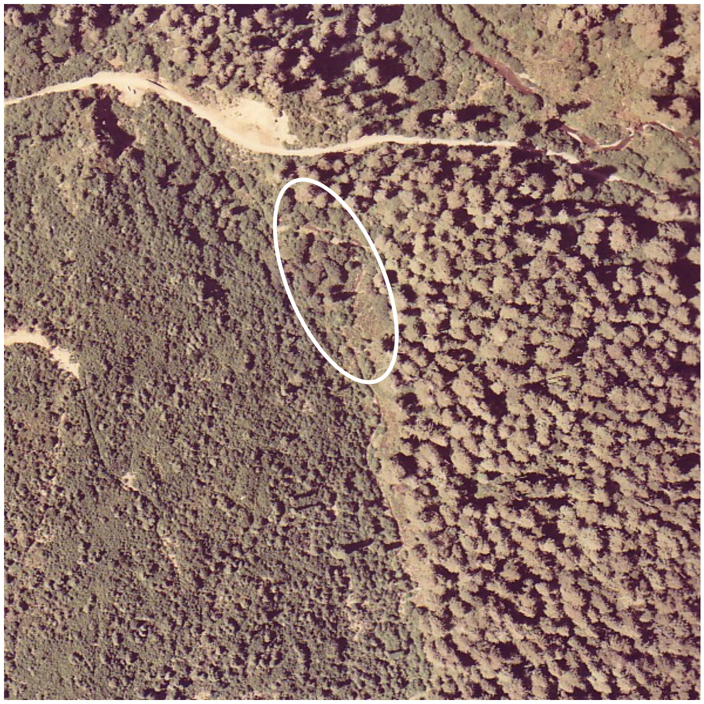
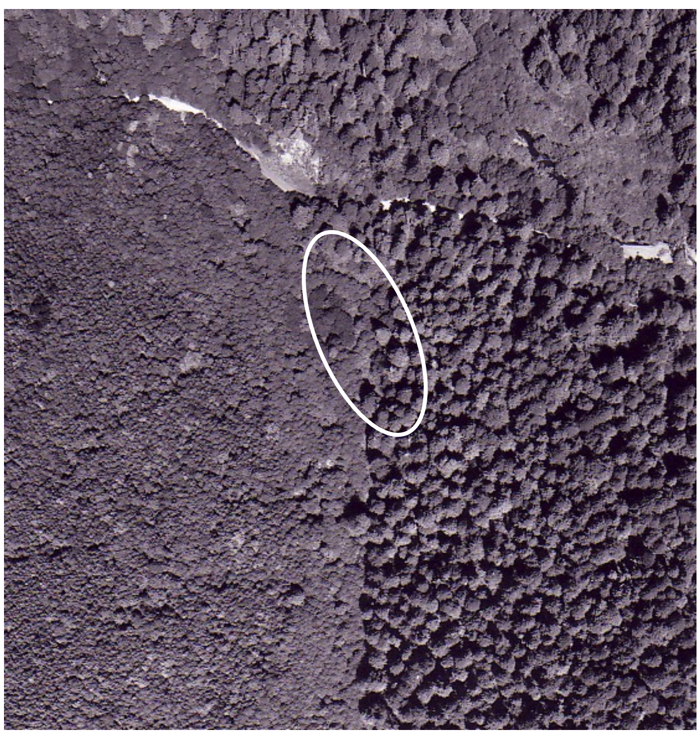


Figure 2-6k. Air photo comparisons of Larry Damm Creek in 1978 and 1997.



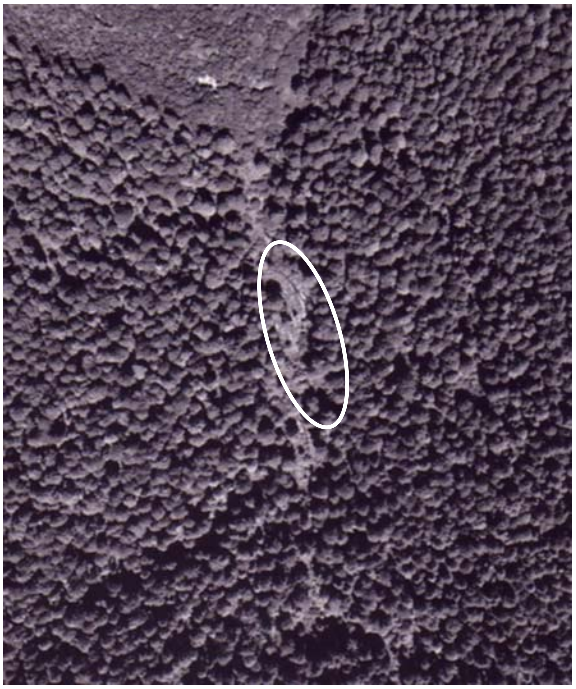
1978



1997

Figure 2-6l. Air photo comparisons of Little Lost Man Creek at the bridge in 1978 and 1997.

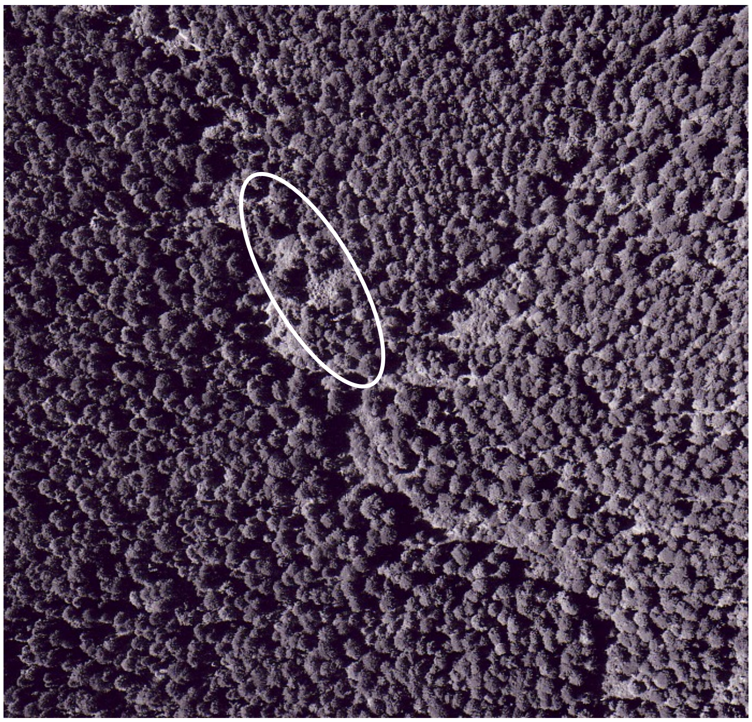
1978



1997

Figure 2-6m. Air photo comparisons of Little Lost Man Creek at the gage in 1978 and 1997.

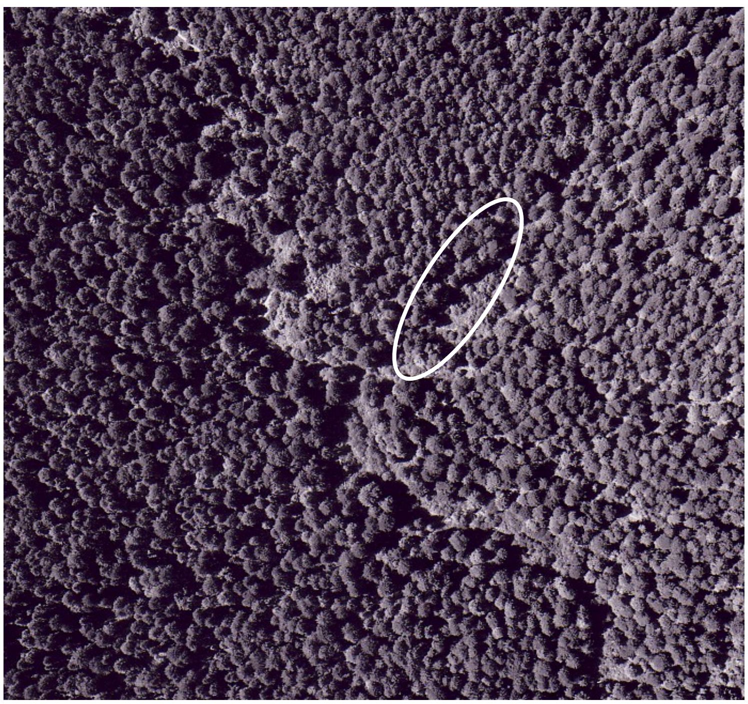
1978



1997

Figure 2-6n. Air photo comparisons of Lost Man Creek downstream of North Fork Lost Man Creek in 1978 and 1997.

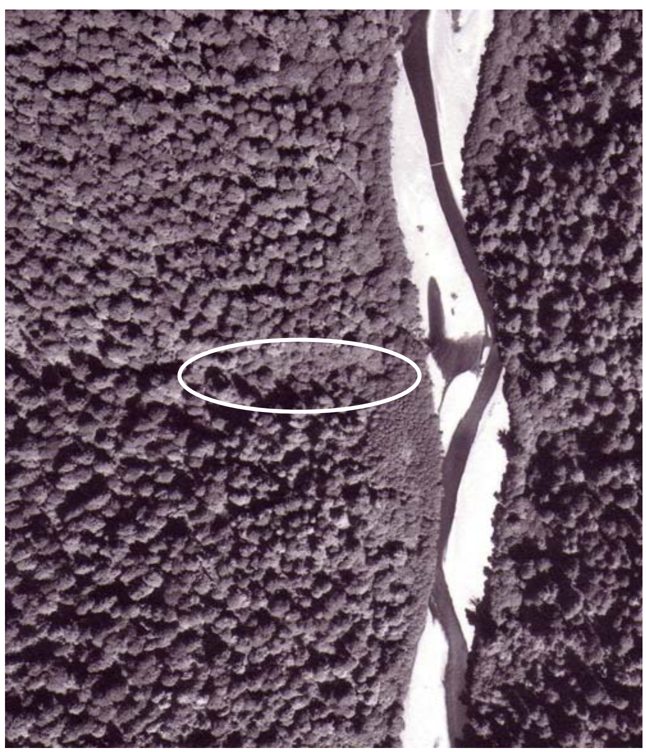
1978



1997

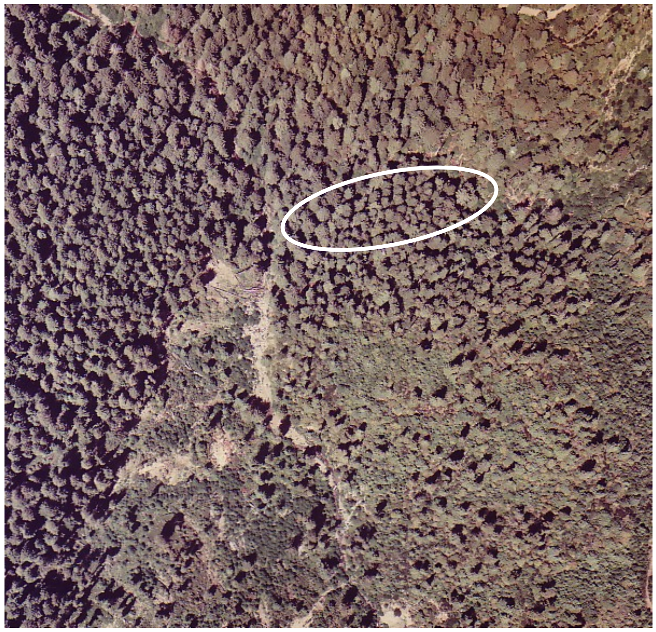
Figure 2-6o. Air photo comparisons of North Fork Lost Man Creek in 1978 and 1997.

1978

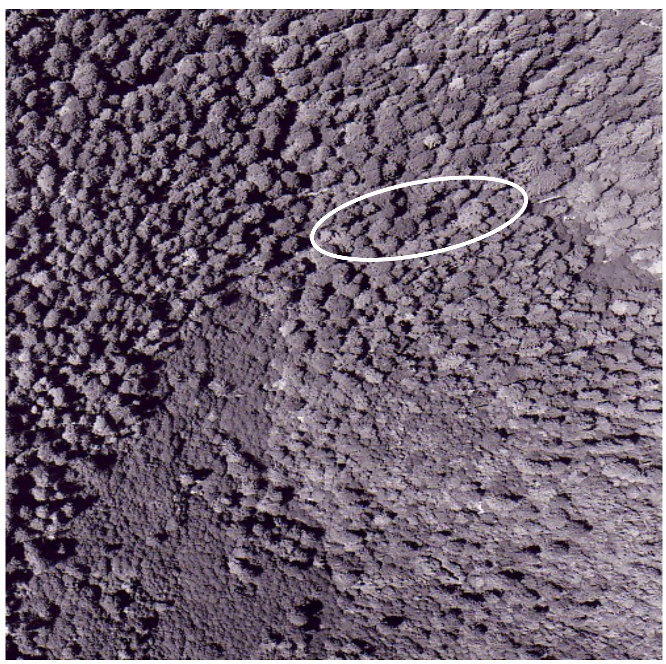


1997

Figure 2-6p. Air photo comparisons of McArthur Creek in 1978 and 1997.



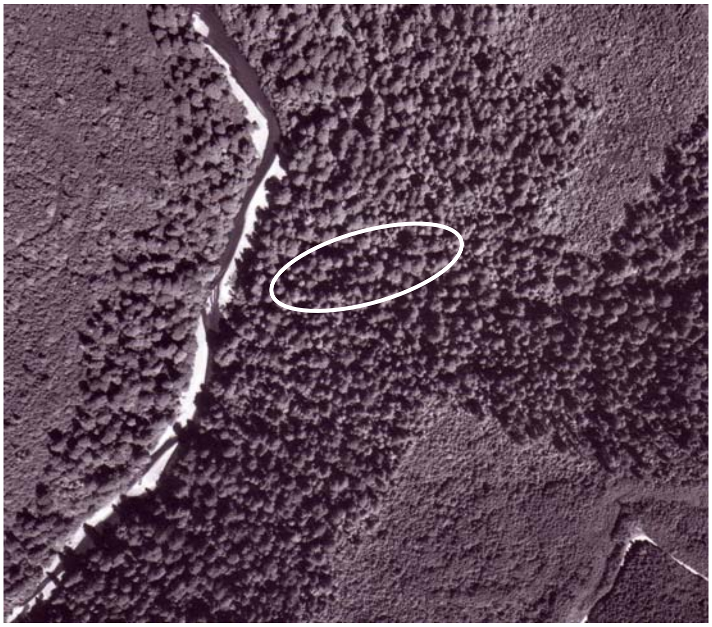
1978



1997

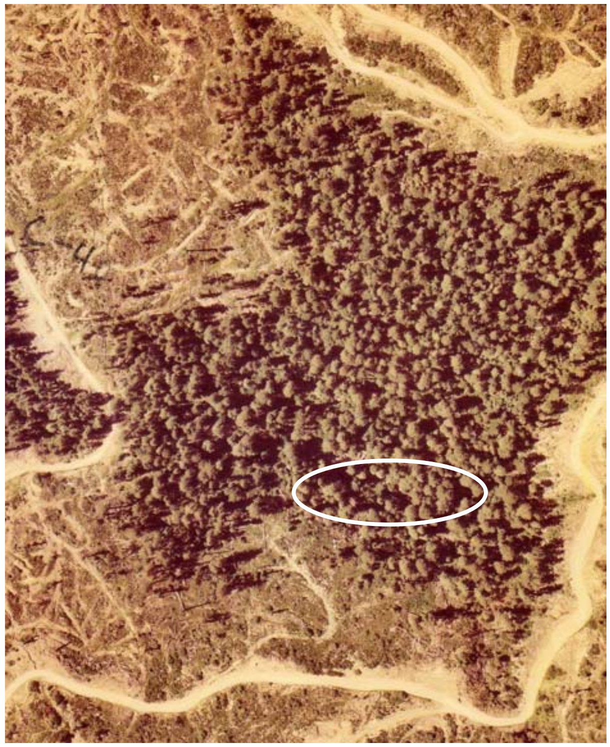
Figure 2-6q. Air photo comparisons of Middle Fork Lost Man Creek in 1978 and 1997.

1978



1997

Figure 2-6r. Air photo comparisons of Lower Miller Creek in 1978 and 1997.


1978

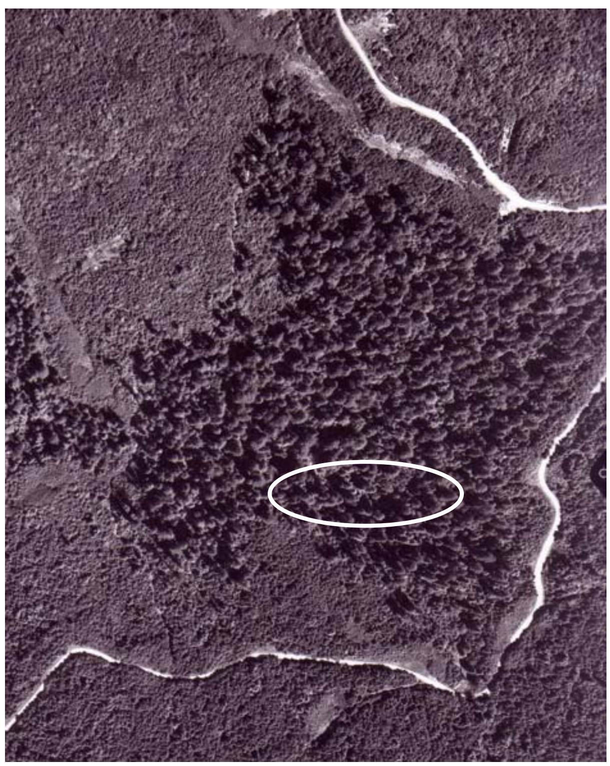
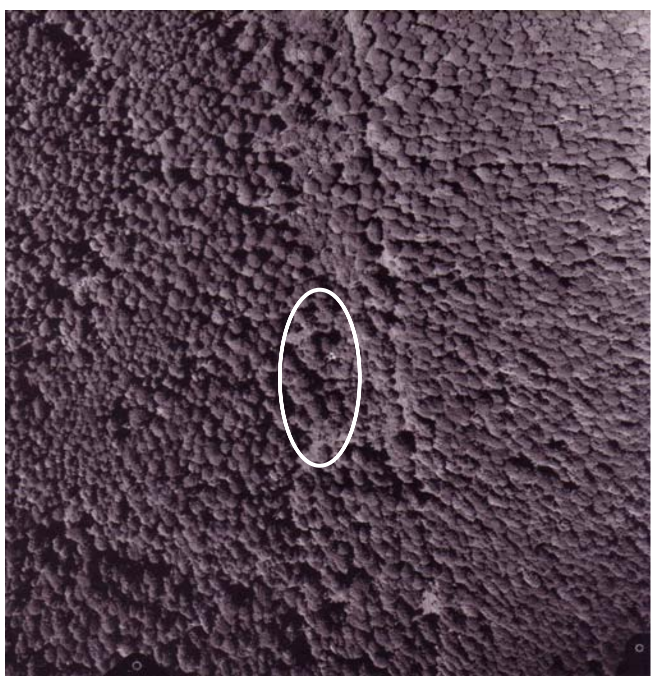

1997

Figure 2-6s. Air photo comparisons of Upper Miller Creek in 1978 and 1997.

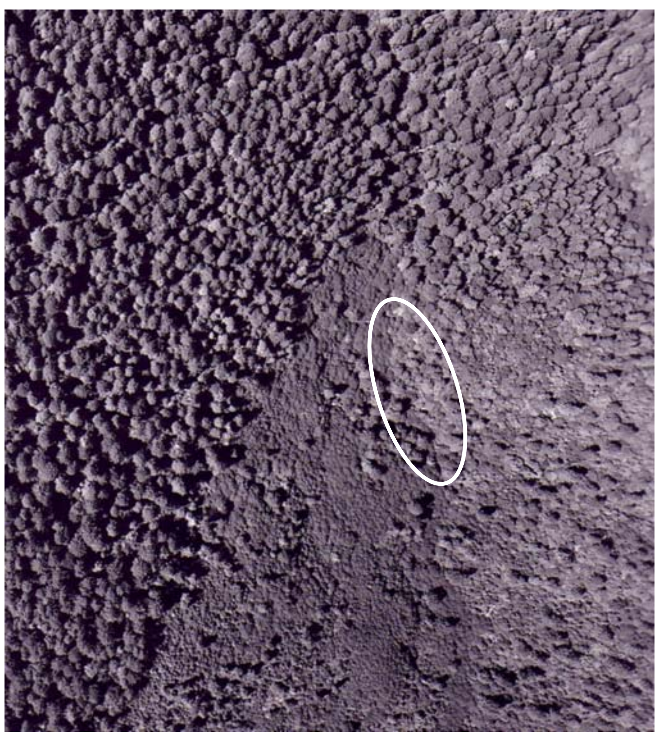
1978



1997

Figure 2-6t. Air photo comparisons of Upper Prairie Creek in 1978 and 1997.

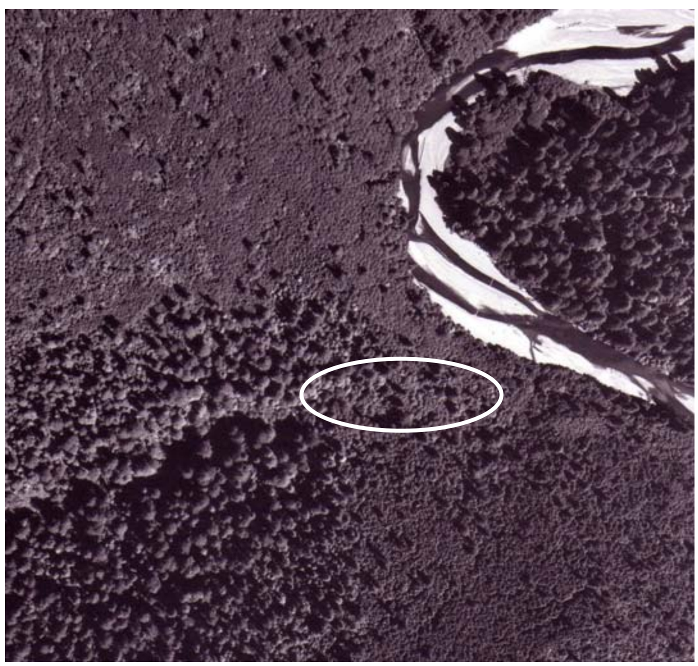
1978



1997

Figure 2-6u. Air photo comparisons of South Fork Lost Man Creek in 1978 and 1997.

1978



1997

Figure 2-6v. Air photo comparisons of Tom McDonald Creek in 1978 and 1997.

Chapter 3 Periphyton

Introduction

Periphyton communities in lotic waters are useful indicators of stream health and are influenced by sunlight, discharge, water nutrient concentrations, sedimentation, temperature and grazing pressure. Periphyton is an assemblage of organisms including benthic algae, detritus, microbes and microzoans held within a polysaccharide matrix secreted by the microorganisms themselves (Lock and others, 1984). Autotrophic production by periphyton provides a substantial food source for higher trophic levels in stream ecosystems through the quality of food that they are able to provide through photosynthesis. Because periphyton is an important component of the food web, it is useful to examine changes in periphyton production in relation to changes in land use. Benthic periphyton are useful indicators of water quality due to their sessile nature and short life cycles (Lowe and Laliberte, 1996) which result in a rapid response to shifts in environmental conditions. Due to their inability to move away from undesirable environmental influences, they must either tolerate the disturbance or die.

Timber harvest, especially within the riparian zone, can lead to an increase in disturbance in these systems, including but not limited to changes in riparian canopy cover, water temperature, sedimentation and water nutrient concentrations. Road removal involving road-stream crossing excavations can also create short-term disturbances in the adjacent stream channels. Various studies have examined the importance of such land use disturbances in affecting the distribution and biomass of periphyton (Lyford and Gregory 1975; Iwatsubo and Averett 1976; Gregory 1980, Murphy and Hall 1981; Noel and others, 1986; Vis and others, 1998; Leland, 1995; Parkhill and Gulliver, 2002). Periphyton biomass was found to be higher in open stream sections of the Pacific Northwest that had been clearcut as opposed to those shaded by riparian canopy (Lyford and Gregory,1975; Gregory 1980; Murphy and Hall, 1981), although periphyton biomass could not be directly related to past land use histories of tree harvesting in other studies (Batzer and others, 2000). Parkhill and Gulliver (2002) experimentally added sediment to outdoor experiment stations and found that periphyton biomass as reported by chlorophyll *a* was significantly lower in treated streams. Grazing pressure by aquatic organisms can also affect periphyton biomass (Gregory 1980; Feminella and others, 1989). Gregory (1980) found that laboratory streams with high and intermediate grazing resulted in decreases in periphyton biomass.

The objectives of this study were to: 1) collect periphyton samples following the same protocol used by the USGS in 1973-75 and compare the biomass to results found in 1973-75 and 2) to evaluate the effects of differing degrees of road rehabilitation and watershed restoration on periphyton biomass.

Methods

In order to compare current periphyton biomass and daily accrual rates with that collected in 1973-75, periphyton ash-free dry mass (AFDM) was measured during the spring and summer of 2004 and 2005 to replicate previous sampling by the USGS. In 2004, six sites were sampled in the spring and nine sites in the late summer, and in 2005 12 sites were sampled in the spring and 11 sites in the late summer. Plexiglass plates cut into 10 x 10 cm squares were placed in the chosen study reaches and allowed to colonize for approximately 60 days. Two plates each were nailed into the stream bed at three randomly chosen riffles per reach for a total of six plates per site. The plates were installed parallel to the stream bed using 15-cm nails. Spring sampling plates were placed in the sites in late May through June and late summer sampling plates were placed in the sites in mid-August through early September. Dates were chosen to correspond with previous measurements made by the USGS in 1973-75. Water velocity and canopy measurements were taken at each plate. After the colonization period, plates were scraped with a plastic bristled brush to remove periphyton and rinsed with stream water filtered through a PUR® water filter into a collecting tray. Periphyton scraped from the two plates in each riffle was composited for a total of one sample per riffle. The sample was filtered through a mesh net into a graduated cylinder to remove debris, mixed well and filtered onto pre-combusted 25 µm glass fiber filters for AFDM determination, with volumes of water and sample noted. Filters were kept frozen until time of analysis. Filters were oven-dried at 100° C for 24 hours, allowed to cool in a desiccator and weighed on an analytical balance to the nearest 0.0001g to obtain a dried mass. The filters were then ashed at 500° C for 1 hour, allowed to cool in a desiccator, rewet with a few drops of distilled water to restore the waters of hydration, oven-dried at 100° C for 24 hours and weighed to obtain an ashed weight (Steinman and Lamberti, 1996).

Results

In order to estimate daily rates of periphyton accrual over the study period biomass of periphyton was determined from each site (Table 3-1). The biomass was divided by the number of days the plates were allowed to colonize in order to obtain daily rates. Both the organic (periphyton) and inorganic mass of the deposited periphyton were calculated. Periphyton accrual rates were generally much lower in the spring and summer of 2004 and 2005 compared to accrual rates in the spring and summer of 1974 and 1975.

With the exception of Bridge Creek, spring accrual rates were higher in the 1970's than in 2004 or 2005 (Figure 3-1). Spring accrual rates over the period of both studies were the highest in Lower Miller Creek (0.07 $(g/m^2)/d$) in the spring of 1974 and were the lowest (0.0 $(g/m^2)/d$) in Hayes Creek in 1974 and Upper Miller Creek in 1975. The high rate at Lower Miller Creek in 1974 was probably related to bridge construction upstream of the site (Averett and Iwatsubo, 1995). During the 2004-2005 spring sampling period, accrual rates were lower in 2005 for every site with the exception of Hayes Creek, an unlogged site (Figure 3-2).

Accrual rates for each site in the summer of 1974 were consistently higher than rates in the summer of 2004 and 2005 (Figure 3-3). In 1974 and 2005, summer accrual rates were higher for every site than spring accrual rates, although this trend was not seen in 2004. Previous studies did not measure summer periphyton accrual rates in 1975. In 1974, summer accrual rates were the highest (0.14 $(g/m^2)/d$)) in Bridge Creek and the lowest in Upper Miller Creek (0.01 $(g/m^2)/d$)). During the 2004-2005 summer sampling period, Lower Miller Creek, an old disturbance site, had the highest (0.018 $(g/m^2)/d$)) accrual rate in 2005 and the lowest (0.001 $(g/m^2)/d$)) in 2004 (Figure 3- 4).

Table 3-1. Rates of periphyton accrual for selected tributaries of Redwood Creek.

									Periphyton	
Site	Date Sampler Installed	Temperature °C	Time	Date Sampler Retrieved	Temperature °C	Time	Colonization Period (days)	Dry Accrual Rate $(g/m^2)/d$	Inorganic Accrual Rate $(g/m^2)/d$	Organic Accrual Rate $(g/m^2)/d$
Bridge Creek	5/13/1974	na	na	7/15/1974	na	na	63	0.02	0.01	0.01
(BRI)	7/15/1974	na	na	9/16/1974	16.0	1200	63	0.43	0.29	0.14
	6/7/1975	na	na	7/31/1975	na	na	54	0.59	0.54	0.05
	6/22/2004	12.5	1400	9/1/2004	15.4	1200	72	0.043	0.026	0.017
	9/1/2004	15.4	1200	10/21/2004	11.0	1200	51	0.036	0.02	0.016
	6/6/2005	10.8	1200	8/4/2005	15.4	1200	60	0.017	0.013	0.004
	8/4/2005	15.4	1200	10/4/2005	11.7	1200	62	0.041	0.028	0.013
Harry Weir	5/13/1974	10.0	1550	7/15/1974	na	na	63	0.1	0.07	0.03
(EMR)	7/15/1974	na	na	9/16/1974	13.0	1330	63	0.12	0.09	0.03
	6/1/1975	12.5	1410	7/31/1975	12.5	1130	60	0.2	0.18	0.02
	6/22/2004	12.5	1120	9/1/2004	na	na	72	0.017	0.011	0.006
	9/1/2004	na	na	10/21/2004	10.5	1325	51	0.01	0.006	0.004
	6/1/2005	12.1	1030	8/1/2005	14.2	1243	62	0.011	0.008	0.002
	8/1/2005	14.2	1243	10/4/2005	10.8	1243	65	0.03	0.023	0.007
Tom McDonald	6/23/2004	12.4	1200	8/31/2004	14.5	1200	70	0.023	0.013	0.01
(TMC)	8/31/2004	14.5	1200	10/21/2004	10.0	1000	52	0.022	0.014	0.008
	6/2/2005	10.8	1200	8/1/2005	13.9	1200	61	0.008	0.006	0.002
	8/1/2005	13.9	1200	10/4/2005	10.5	1440	65	0.031	0.022	0.009

Table 3-1. continued

Site	Date Sampler Installed	Temperature °C	Time	Date Sampler Retrieved	Temperature °C	Time	Colonization Period (days)	Dry Accrual Rate (g/m²)/d	Periphyton Inorganic Accrual Rate (g/m²)/d	Organic Accrual Rate (g/m²)/d
Lower Miller	5/14/1974	9.0	1410	7/16/1974	na	na	63	1.19	1.12	0.07
(MLRL)	7/16/1974	na	na	9/17/1974	12.0	1200	63	0.92	0.85	0.07
	5/31/1975	12.0	1150	7/28/1975	na	na	58	0.03	0.02	0.01
	6/15/2004	na	na	8/23/2004	na	na	70	0.017	0.011	0.006
	8/23/2004	na	na	10/29/2004	9.9	1040	68	0.002	0.001	0.001
	6/8/2005	na	na	8/8/2005	13.5	1120	62	0.025	0.02	0.005
	8/8/2005	13.5	1120	10/6/2005	10.0	1115	61	0.102	0.084	0.018
Fortyfour	6/14/2004	12.0	1200	8/23/2004	na	na	71	0.02	0.01	0.01
(FOR)	8/23/2004	na	na	10/20/2004	10.9	1100	59	0.007	0.003	0.004
	6/8/2005	na	na	8/8/2005	13.5	1254	62	0.02	0.017	0.003
	8/8/2005	13.5	1254	10/6/2005	10.0	954	61	0.062	0.046	0.016
Hayes	5/15/1974	9.5	1530	7/15/1974	na	na	61	0.06	0.06	0
(HAY)	7/15/1974	na	na	9/14/1974	12.5	1400	61	0.01	0.01	0
	6/17/2004	11.5	1115	8/26/2004	na	na	71	0.014	0.008	0.006
	5/24/2005	10.0	830	7/20/2005	13.0	910	58	0.086	0.058	0.028
Lost Man	5/10/1974	11.5	1305	7/15/1974	na	na	66	0.05	0.03	0.02
below North	7/15/1974	na	na	9/15/1974	14.0	1100	62	0.12	0.08	0.04
Fork (LM3)	6/2/1975	12.0	1130	7/27/1975	na	na	55	0.05	0.03	0.02
	8/20/2004	15.4	1200	10/19/2004	na	na	61	0.038	0.023	0.015
	5/25/2005	11.1	1130	7/22/2005	14.3	1200	59	0.022	0.018	0.004
	7/22/2005	14.3	1200	9/19/2005	12.6	1200	60	0.041	0.03	0.011

Table 3-1. continued

Site	Date Sampler Installed	Temperature °C	Time	Date Sampler Retrieved	Temperature °C	Time	Colonization Period (days)	Periphyton Dry Accrual Rate (g/m^2)/d	Periphyton Inorganic Accrual Rate (g/m^2)/d	Periphyton Organic Accrual Rate (g/m^2)/d
Little Lost	8/20/2004	na	na	10/18/2004	12.0	1120	60	0.039	0.023	0.017
Man at bridge	5/24/2005	11.0	1300	7/20/2005	14.0	1210	58	0.034	0.028	0.006
(LLML)	7/20/2005	14.0	1210	9/19/2005	12.0	1057	62	0.032	0.023	0.009
Little Lost	7/15/1974	na	na	9/14/1974	na	na	61	0.46	0.35	0.11
Man at gage	6/2/1975	11.5	1400	7/27/1975	na	na	55	0.01	0.06	0.04
(LLM)	8/20/2004	15.1	1200	10/18/2004	na	na	60	0.037	0.026	0.011
	5/24/2005	na	na	7/20/2005	14.3	1200	58	0.039	0.031	0.008
	7/20/2005	14.3	1200	9/19/2005	11.8	1200	62	0.042	0.032	0.01
Larry Damm	8/20/2004	na	na	10/19/2004	11.0	915	61	0.01	0.006	0.004
(LDC)	5/24/2005	11.0	1330	7/20/2005	14.0	1310	58	0.006	0.005	0.001
	7/20/2005	14.0	1310	9/19/2005	11.0	1250	62	0.007	0.005	0.002
Godwood	5/25/2005	10.1	930	7/22/2005	12.5	925	59	0.021	0.017	0.004
(GOD)	7/22/2005	12.5	925	9/20/2005	10.5	859	61	0.035	0.026	0.008
Upper Miller	5/31/2005	10.9	1036	7/28/2005	12.5	900	59	0.027	0.023	0.004
(MLRU)	7/28/2005	12.5	900	9/23/2005	10.5	925	58	0.04	0.033	0.006

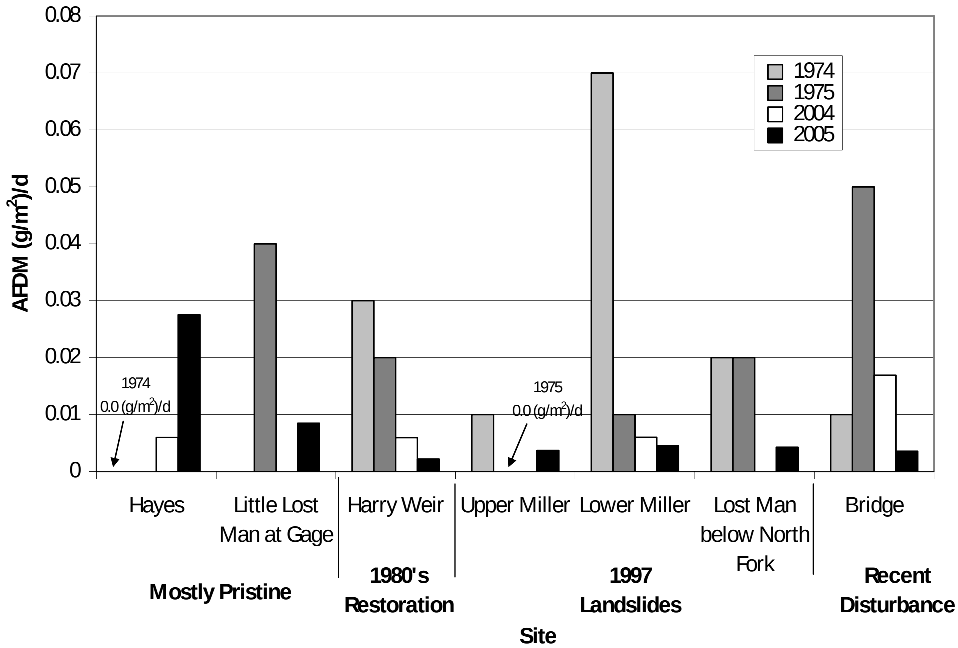


Figure 3-1. Spring periphyton accrual rates for selected tributaries of Redwood Creek sampled in 1974, 1975, 2004 or 2005. Hayes and Upper Miller Creeks had values of 0.00 $(g/m^2)/d$ in spring 1974 and 1975, respectively. There are no available data for all other missing data points.

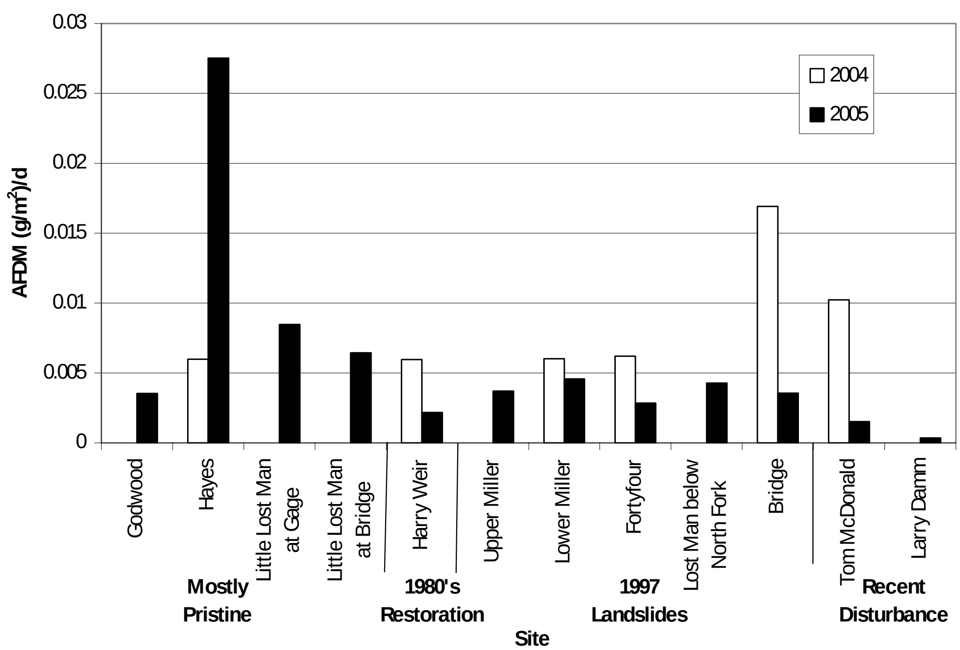


Figure 3-2. Spring periphyton accrual rates for selected tributaries of Redwood Creek sampled during 2004 or 2005. There are no available data for missing data points.

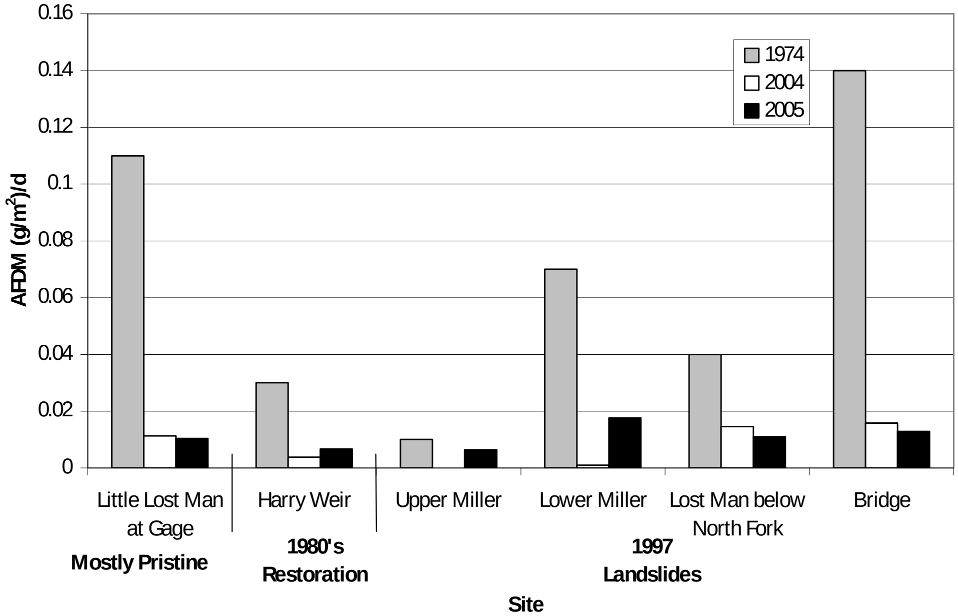


Figure 3-3. Summer periphyton accrual rates for selected tributaries of Redwood Creek sampled during 1974, 2004 or 2005. There are no available data for missing data points.

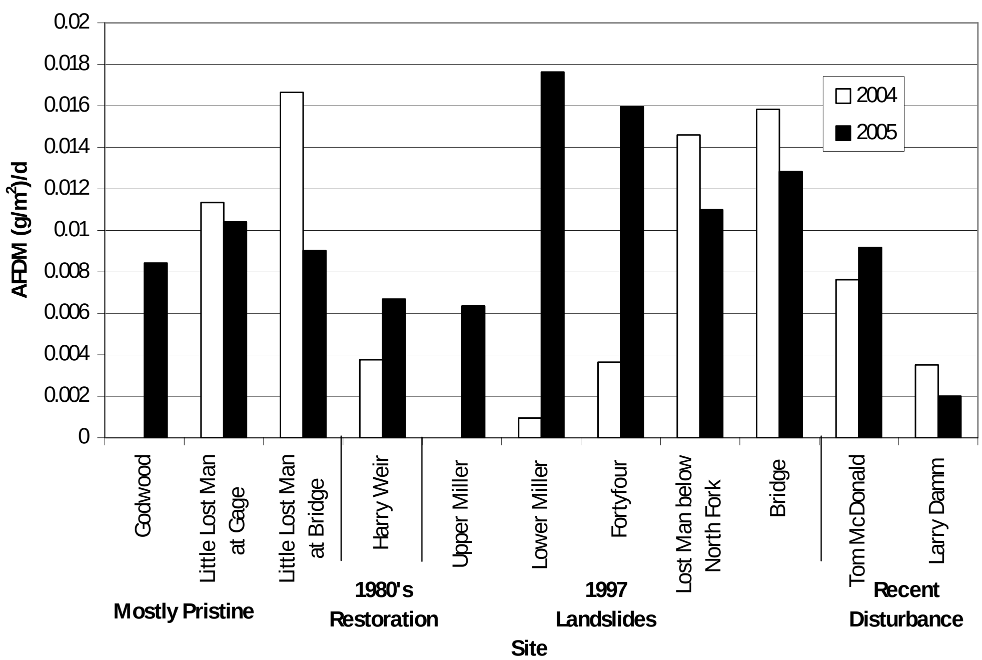


Figure 3-4. Summer periphyton accrual rates for selected tributaries of Redwood Creek sampled during 2004 or 2005. There are no available data for missing data points.

Canopy cover influences periphyton growth because it limits light reaching the channel bed, and an increase in canopy cover decreases the amount of light available for in-stream primary productivity. Percentage of canopy cover has increased in every tributary sampled in 2005 when compared to cover in the early 1970's (Figure 3-5). Sequential aerial photographs also show increasing canopy cover on stream channels (Figures 2-6a – 2-6v). This increase has been most pronounced at Bridge Creek where the percentage of canopy cover increased from 3 percent in the 1970's to 87 percent in 2005.

Nutrient levels were not measured in the 2004-2005 study, although in the 1970's nitrogen and phosphorus levels were low (Iwatsubo and others, 1975). These low levels were still high enough to allow for modest periphyton growth in both the spring and summer, therefore in the current study we did not measure nutrient levels as one of the factors contributing to periphyton growth in the study tributaries. Periphyton growth was faster in the summers of 1974 and 2005 than in the spring, which is expected due to increased water temperatures in the summer months (Table 2-3). Grazing pressure, as measured by percentage of scrapers, does not appear to be affecting the low periphyton accrual rates in either the spring or summer of 2004 and 2005 (Figure 3- 6).

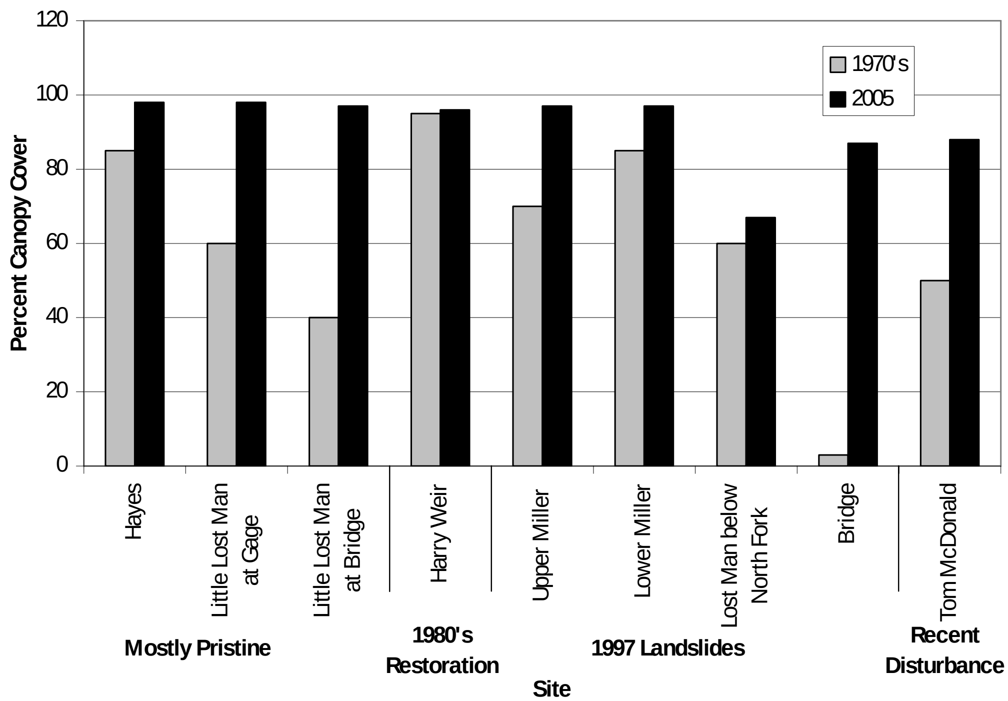


Figure 3-5. Mean percentage of canopy cover for selected tributaries of Redwood Creek in both the 1973-1975 and 2004-2005 studies.

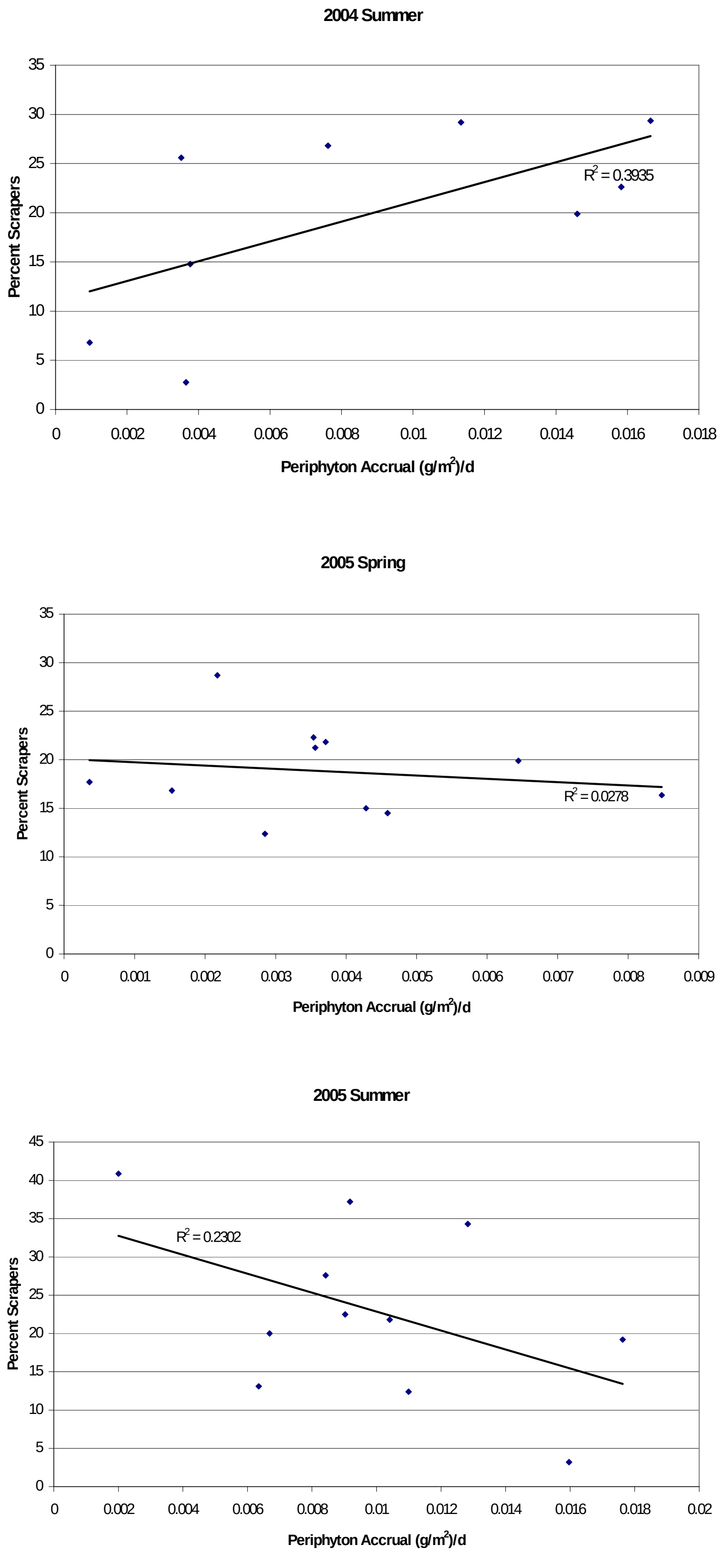


Figure 3-6. Mean daily periphyton accrual rates for selected tributaries of Redwood Creek vs. percentage of scrapers collected with a 500 µm mesh D-frame benthic kick net.

Discussion and Conclusions

Periphyton growth measured in 2004-2005 was lower in almost every stream than that measured in the early 1970's. This was probably due to the increase in canopy cover at the sampling sites. Canopy cover consists of both hardwoods (alder) and conifers (redwood and Douglas fir), but by late spring and summer both types of vegetation are effective in shading these small streams. Water discharge also affects periphyton growth. A large flood in 1975 resulted in severe channel aggradation and widening in Bridge Creek (Figure 2-2). The lack of canopy cover at that time probably caused the high periphyton accrual rates measured there in late spring. Higher spring flows in 2005 than in 2004 probably account for the lower spring accrual rates at most sites in 2005.

Benthic algae may enter the food web through consumption by benthic invertebrates, such as snails, and consequently periphyton biomass may be influenced by the abundance of scrapers. In this study, however, there was no clear relationship between percentage of scrapers and periphyton biomass.

Because most of the sampled sites had similar values for canopy cover in 2004 and 2005, and nutrient differences are negligible (Averett and Iwatsubo, 1995) differences in upstream land use were not apparent in periphyton growth rates. There does not appear to be a relationship between watershed disturbance level and spring or summer accrual rates. The lack of any clear trends in periphyton biomass, as measured by AFDM, makes it difficult to draw any conclusions concerning the response of periphyton to the degree of road rehabilitation and watershed disturbance. The reestablishment of riparian shading on the stream channels seems to be the dominant factor in controlling periphyton accrual rates.

Chapter 4 Macroinvertebrates

Introduction

Aquatic macroinvertebrates serve as the major link between primary producers and organisms belonging to higher trophic systems. Rosenberg and Resh (1993) have provided a summary of advantages and disadvantages to consider when using benthic macroinvertebrates as biological indicators. Benthic macroinvertebrates are advantageous indicators because by their ubiquitous nature they are affected by all types of water and habitats, their large numbers of species offer a wide range of responses, their sedentary nature allows spatial analysis of disturbances and their long life cycles allow temporal changes of regular or intermittent occurrences to be examined. However, they can be difficult to monitor because they do not respond to all types of disturbance, factors other than water quality can affect their distribution and abundance, seasonal variation and macroinvertebrate drift may complicate interpretations and quantitative sampling requires large number of samples which can be time consuming and expensive to analyze.

Many studies have used benthic macroinvertebrate communities as indicators of the level of disturbance to aquatic systems (Iwatsubo and Averett, 1976; Erman and others, 1977; Newbold and others, 1980; Gurtz and Wallace, 1984; Robinson and Minshall 1986; Noel and others, 1986; Fore and others, 1996; Hetrick and others, 1998; Barbosa and others, 2001). Erman and others (1977) found that benthic invertebrate communities from streams logged without protective measures in northern California were significantly different from communities from unlogged streams and that a direct correlation existed between increases in diversity index and increases in buffer width. Streams with bufferstrips wider than 30 meters did not display logging impacts. In the same study Newbold and others (1980) reported that densities of total macroinvertebrate fauna and of Chironomidae, Baetis and Nemoura were higher in unprotected streams than in controls. Little effect was seen on community richness and diversity after Ciesielka and Bailey (2001) distributed 6.6L of silt along a stretch of a 4th order stream to disturb benthic macroinvertebrates after a period of either three hours or three days.

Recent studies using benthic macroinvertebrates as biological indicators have employed rapid bioassessment protocols based on a multimetric approach to evaluate water quality (Resh and others, 2000; Morley and others, 2002; Metzeling and others, 2003). This approach was first introduced in 1981 with the index of biological integrity, or IBI, which used a range of attributes of fish assemblages as biological indicators (Karr, 1981). Different physical features of a stream are assessed and benthic macroinvertebrates are collected and identified and this information is used to calculate a biotic index score, based on the premise that disturbance tolerances differ among various organisms (Resh and others, 1996). Organisms are assigned values according to their degree of tolerance to a pollutant. The tolerance of the taxa that comprise a particular community and the numbers of each taxon are used to calculate a single score. These scores help to determine the biological integrity and pollution status of a system. The key to building an effective multimetric index is finding attributes that change in response to human impact and metrics that are able to discriminate human-caused changes from the background noise of natural variation (Karr and Chu, 1999). Rapid bioassessment approachs are popular because they limit the number of habitats examined, the number of samples collected, the amount of sample sorting time and the number of taxonomic identifications to be made making them much more cost effective (Rosenberg and Resh, 1996).

Functional feeding group, or FFG, classification was first introduced by Cummins (1973) and is based on the morphobehavioral mechanisms of food acquisition. The macroinvertebrate functional feeding group classification is based on the idea organisms evolved certain morphological-behavioral food-gathering mechanisms or locomotion-attachment adaptations and can be placed into particular groups based on these mechanisms (Merritt and Cummins, 1996). The general functional group categories are: shredders, which feed on coarse particulate organic matter (particles greater than 1mm in size), collectors, which feed on fine particulate organic matter (particles less than 1mm and greater than 0.5 µm in size), scrapers, which feed on periphyton and predators, which feed on prey. The functional feeding group approach is advantageous because it allows an assessment, numerically or by standing crop biomass, of the degree to which the invertebrate biota of a given aquatic system is dependent upon a particular food resource (Merritt and Cummins, 1996). This categorization reflects 1) the biochemical differences in the nutritional resources and 2) whether the major source of the food is autochthonous (produced from within the aquatic system) or allochthonous (produced from the stream-side or riparian terrestrial area) (Merritt and Cummins, 1996).

Several studies have attempted to identify the aquatic macroinvertebrate communities of the Redwood Creek watershed (Averett and Iwatsubo, 1981; Harrington 1983; Anderson 1988). Averett and Iwatsubo (1995) concluded that the numbers and types of benthic invertebrates were directly related to the type of bed material. The objectives of the present study were to: 1) collect benthic macroinvertebrate samples and compare the results with the data collected in 1973-75 2) collect macroinvertebrate data following a rapid bioassessment protocol to evaluate the effects of differing stages of road rehabilitation and land use on streams 3) provide a current list of benthic macroinvertebrates with associated metrics from the Redwood Creek basin.

Methods

Field Sampling

Benthic macroinvertebrates were sampled using two methods in both the spring and late summer of 2004 and 2005 within selected tributaries of the Redwood Creek watershed basin. Surber samples were collected from a total of ten tributaries during the spring of 2004 and nineteen tributaries during the late summer of 2004. Benthic kick net samples were collected in the late summer of 2004 from a total of 19 tributaries. Surber samples were collected from two tributaries during the spring of 2005 and benthic kick net samples were collected from 22 sites in the spring and 20 sites in the late summer of 2005. All benthic samples were collected from three randomly (n =3) selected riffles within a 100 meter designated study reach within each tributary sampled. Surber samples were collected with a 250 µm mesh net by disturbing a 0.30 m^2 area of the riffle substrate for approximately one minute. Invertebrates were carefully removed from the Surber sampler and washed into a 250 µm sieve. Benthic organisms were then put into whirl-pak® bags and preserved in 75% ethanol for later identification.

Benthic kick net samples were collected according to the California Stream Bioassessment Procedure (Harrington and Born, 2000), a regional adaptation of the United States Environmental Protection Agency's Rapid Bioassessment Protocol. Using this method a total of three timed kicks, each covering a 0.3 x 0.6 m portion of the substrate upstream of the net, were performed across the width of the riffle channel using a 500 µm mesh D-frame kick net. These three kicks were combined to become one of three replicate samples collected within the 100 meter study reach. Benthic organisms retained in the net were removed and treated in the same manner previously described.

Laboratory Methods

Two different methods were used to sub-sample benthic organisms collected using the Surber sampler and the D-frame kick net. Surber samples (n = 3) collected from each site during 2004 (n_{spring} = 10 and n_{fall}= 19) were subsampled using an invertebrate sample splitter. The sample splitter was used to divide each of the benthic samples into two equal fractions. One of the three benthic samples was then selected at random. Benthic organisms from both fractions of the sample were then identified to the family level (in most cases). A Chi-square goodness-of -fit analysis (p = 0.05) was then performed on each fraction of the sample to ensure that number of individuals within each taxonomic category was representative of the sample as a whole. If there were no significant differences calculated in the number of organisms within each taxonomic category for a site, then one fraction from the remaining two samples was randomly selected for identification. The two Surber samples collected in 2005 were not split. Estimates of functional feeding group community structure, benthic density, diversity and taxonomic distribution were averaged over the three riffles and then compared to similar analyses completed in the early 1970's (Iwatsubo and others, 1975 and 1976).

Benthic kick net samples were subsampled according to those methods outlined in the California Stream Bioassessment Procedure (Harrington and Born 2000). The preserved samples from each site (n=3) were cleaned and combined in the laboratory for a total of one sample per study reach. Subsamples from each of these samples were prepared by spreading the sample evenly across a tray marked with 5 x 5cm grids to an approximate thickness of 1 cm. Each grid was removed from the tray with a razor blade and placed in a petri dish containing 85 percent ethanol for laboratory identification. Invertebrates were identified to the genus level when possible. Enough grids were randomly chosen to be subsampled in order to identify 500 macroinvertebrates per site. Biological metrics were calculated for all Surber and benthic kick net samples.

All Surber samples collected in 2004 and 2005 and all benthic kick net samples collected in 2004, as well as half of the benthic kick net samples collected in the spring of 2005, were identified in the Redwood National Park laboratory located at the South Operations Center, Orick, California by two master's-level aquatic biologists. Voucher specimens were prepared for each distinct taxa identified. Specimens were preserved in 75 percent ethanol and placed in a glass vials along with a record of collection and taxonomic identification. Identification of voucher specimens was checked by Dr. Kenneth Cummins of Humboldt State University's Institute of River Ecology. A portion of the benthic kick net samples collected in the spring of 2005 and all of those collected in the late summer of 2005 were identified by Jonathan Lee of Lee Consulting in Arcata, California.

Results

Surber samples

Density

Invertebrate density is defined as the number of organisms in a given area. In order to be consistent with the 1974-1975 USGS study, density is reported as number of individuals/m^2 of channel bed area and is reported to two significant figures. Density is not commonly reported as a macroinvertebrate metric, but in this case it is a useful way to compare both spring and late summer data between the two studies. Density, as with all other metrics calculated from Surber samples, is a mean of the three riffles sampled within each study reach. Densities were variable in both studies, but in most instances were higher in both the spring and summer of 1974 and 1975 than in 2004 and 2005. Interannual variation in densities was high. Water discharge was similar in 1974 and 2004, and in 1975 and 2005 (Figure 2-2), so comparisons are probably more valid for those pairs of years.

Five of the eight sites sampled in the spring of 1974 and 2004 had lower densities in 2004 than 1974, and the largest decreases were in a pristine stream (Little Lost Man Creek) (Figure 4-1). The three sites that showed similar or higher densities from the spring of 1974 to the spring of 2004 were the more recently disturbed sites. All sites, with the exception of Elam Creek, had higher late summer densities in 1974 than in 2004 (Figure 4-2). In 1974 and 1975, late summer densities were much higher in Little Lost Man Creek at the gage than in other tributaries, but in 2004 Tom McDonald Creek and Lost Man Creek below North Fork had the highest late summer densities. Late summer samples had equal or greater densities than did spring samples over all years with two exceptions, Bridge and Harry Weir Creeks. In 2004, density in Bridge Creek decreased by almost 50 percent, from 2900 n/m^2 in the spring to 1400 n/m^2 in late summer. In 1975, Harry Weir Creek density decreased by 10% from the spring to late summer.

Variation among densities was high between the 1974 and 2004 study years, especially in the pristine sites during the spring sampling period. Density decreased 66 percent from 1974 to 2004 at Little Lost Man Creek at the gage and 62 percent at Little Lost Man Creek at the bridge and Hayes Creek. Invertebrate density at Harry Weir Creek, which had road restoration in the 1980's, decreased 78 percent from the spring of 1974 to 2004. Hayes Creek, one of two sites sampled in the spring of 1975, decreased 96 percent from 1975 to 2005. The density at Upper Miller Creek decreased 95 percent from the summer of 1974 to 2004, while densities at both Little Lost Man Creek at the gage and Cloquet Creek decreased 86 percent from the summer of 1974 to 2004.

There did not appear to be any consistent trend among levels of disturbance and densities in 2004 and 2005. In the spring of 2004, the highest density was seen in Bridge Creek (2,400 n/m^2), and in the summer of 2004, the highest density was in Tom McDonald Creek (4,500 n/m^2), both recently disturbed. The lowest density in the spring of 2004 occurred in Fortyfour Creek at 230 n/m^2, disturbed by a landslide in January 1997, and in the late summer of 2004 in Fortyfour and Upper Miller Creeks at 270 n/m^2.

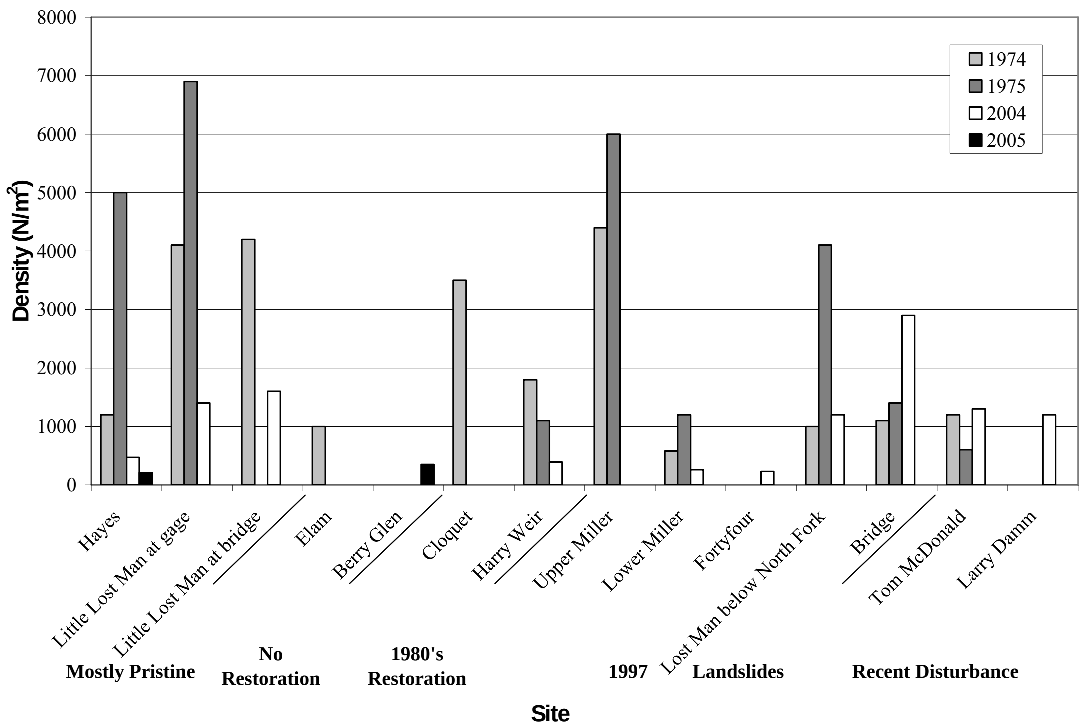


Figure 4-1. Spring macronvertebrate densities sampled from tributaries of Redwood Creek in 1974, 1975, 2004 or 2005 with a 250μm Surber sampler.

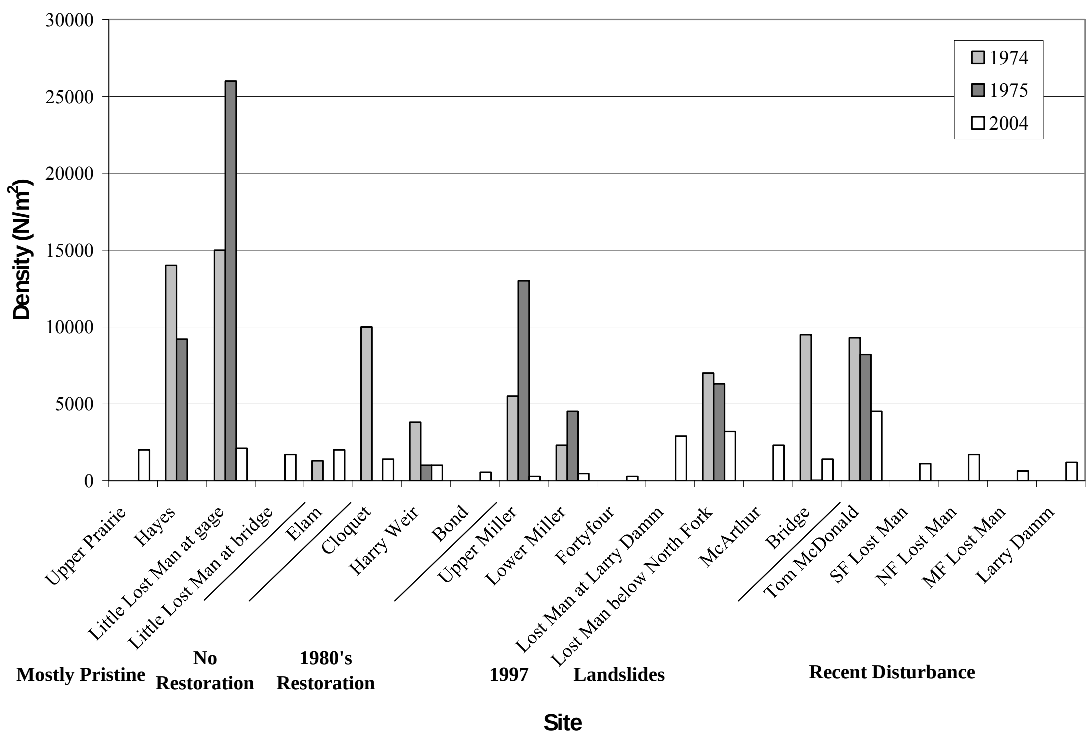


Figure 4-2. Summer macroinvertebrate densities sampled from tributaries of Redwood Creek in 1974, 1975 or 2004 with a 250μm Surber sampler.

Diversity

The increasing diversity of a system correlates with the increasing health of the assemblage and suggests that niche space, habitat and food sources are adequate to support survival and propagation of a variety of species (Harrington and Born, 2000). All diversity measures are based on the general formula that the highest diversity level results from having one individual of many different species, while many individuals of one species will have the lowest value (Harrington and Born, 2000). A diversity index, d, was calculated for each sample collected over all years of the study based on the following equation from Wilhm and Dorris (1968):

$$d = -\sum_{i=1}^{s} n_i/n \log_2 n_i/n$$

where d = diversity index
n_i = number of individuals per taxa
n = total number of individuals and,
s = total number of taxa in the sample of the community

A sample with a low diversity index value indicates that the composition of the sample contains few taxa and a sample with a high diversity index value indicates that the sample contains a large number of taxa.

The benthic invertebrate community consisted of 129 taxa in the 2004 and 2005 sampling seasons. Averett and Iwatsubo (1981) identified 144 taxa in the early 1970's. Different families of invertebrates were identified to different levels of taxonomy in the two studies, so this comparison may not be very appropriate for determining actual differences in numbers of taxa. There were no large differences in taxa between the two studies although since the 1970's there have been several changes in nomenclature. The spring and summer 1974 samples showed higher numbers of Perlodids than were seen in any other year of study. In order to compare diversity from the 1970's study with data collected from 2004-2005, diversity was calculated for both data sets using family as the taxonomic level. Certain taxa that were left at order in both studies, such as Collembola and Oligochaeta, were also counted as distinct taxa. The diversity index for both spring and late summer were compared between the two studies. The diversity index values from spring 2004 were higher than those in the spring of 1974 and 1975 in five of the seven sites sampled in all three years (Figure 4-3). Spring diversity index values were lower in 2004 than 1974 and 1975 in Lower Miller and Tom McDonald Creeks. Bridge Creek had the lowest spring diversity index over all years of study in 1975 (1.33) and Little Lost Man Creek at the gage had the highest in 2004 (3.58). Late summer diversity index values from 2004 were higher than those in 1974 in five of the nine sites sampled in all years (Figure 4-4). The highest late summer diversity index was in 2004 in Bond Creek (3.66) and the lowest was in 1975 in Bridge Creek (1.0).

There was no clear relationship between the diversity index and levels of disturbance. In the spring of 2004 and 2005, the lowest diversity index was Bridge Creek (2.44) in 2004, a recently disturbed site, and the highest was Little Lost Man Creek at the gage (3.4) in 2004, a pristine site. In the late summer of 2004 the highest diversity index was in Bond Creek (3.66), which underwent restoration in the 1980's. The lowest diversity index was Elam Creek (2.0), an old disturbance site that has had no road removal work.

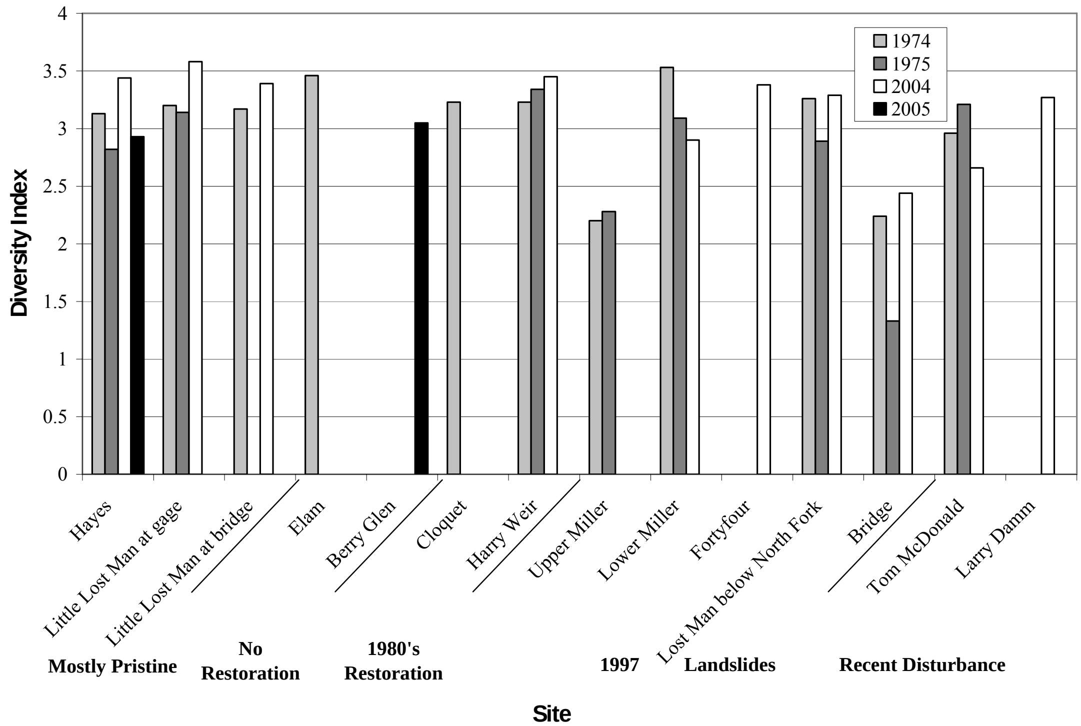


Figure 4-3. Spring macroinvertebrate diversity indexes sampled from tributaries of Redwood Creek in 1974,1975,2004 or 2005 with a 250μm Surber sampler.

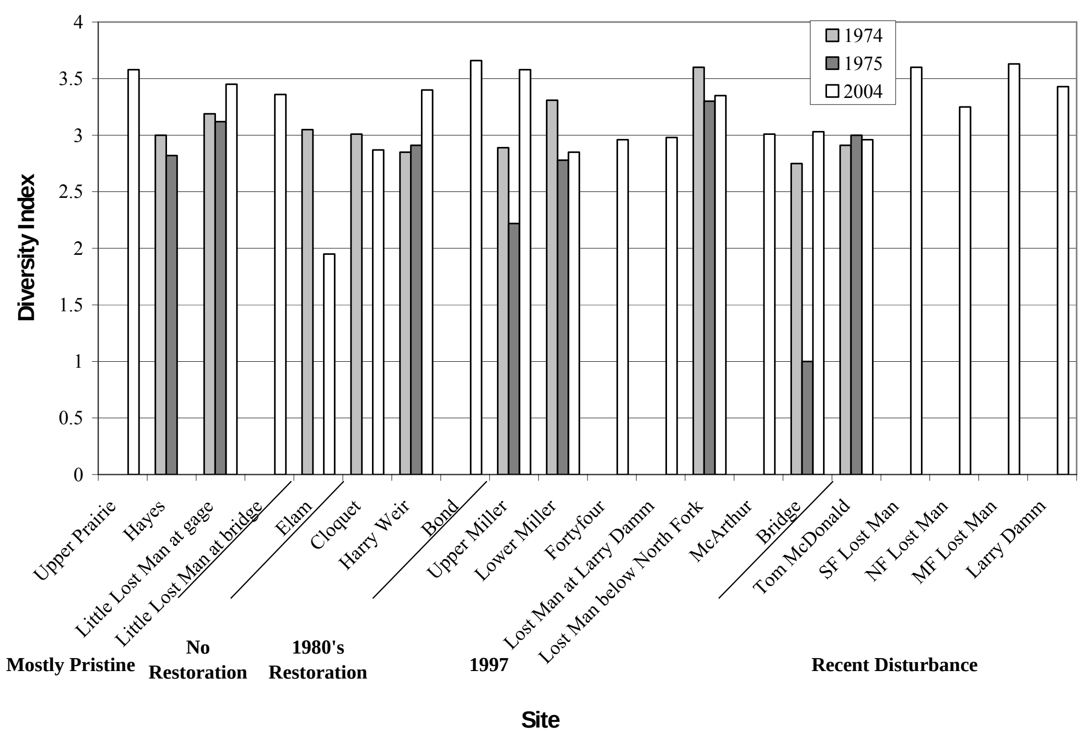


Figure 4-4. Summer macroinvertebrate diversity indexes sampled from tributaries of Redwood Creek in 1974, 1975 or 2004 with a 250μm Surber sampler.

Functional Feeding Groups

Functional feeding group composition (FFG), which is based on the morpho-behavioral mechanisms of food acquisition by aquatic macroinvertebrates, was used to compare the current percentage of composition of FFG's with those found by Averett and Iwatsubo (1981). Percentages of FFG data sets from the 1970's and from 2004 and 2005 were calculated using seven classifications by Merritt and Cummins (1996). Functional feeding group percentages are reported in Appendices 4-1 and 4-2. Each taxon was assigned to one of the following FFG's: shredder, filtering-collector, gathering-collector, predator, scraper, omnivore and piercer-herbivore. Percentages of FFG's from the 1970's data were originally determined using nine functional group categories and therefore had to be recalculated in the present study. Surber samples collected from both spring and late summer in both studies revealed that the gathering-collectors dominated the assemblage. The percentages of scrapers and filtering collectors were higher in the late summer samples than in the spring samples in 2004

Filtering-collectors filter fine particulate matter and are expected to increase in response to disturbance (Harrington and Born, 2000). Percentages of filtering collectors were higher in the spring of 1974 than in 2004 at six sites (Figure 4-5). With the exception of Hayes Creek, spring percentages of filtering-collectors stayed below two at all sites in 2004. Both 2004 and 2005 spring percentages were four-five times higher than in 1974 and 1975 at Hayes Creek, a pristine site. The percentages were higher in both 1974 and 1975 than 2004 in Lower Miller, Lost Man below North Fork and Elam Creeks (Figure 4-6). Lower Miller Creek showed a ten-fold decrease from late summer 1974 to 2004. Percentages of filtering collectors were generally higher over all years in the late summer than in the spring. There was no consistent pattern in the change in percentages of filtering-collectors through time in either spring or summer.

Gathering collectors are the macrobenthos that collect or gather fine particulate matter and are expected to increase in response to disturbance (Harrington and Born, 2000). Late summer percentages of gathering collectors tended to be higher in 1975 than in 2004. Gathering collectors generally dominated the invertebrate community over all years and comprised over 80 percent of the community at Bridge Creek in the spring of 1975 and Upper Miller Creek in the spring of 1974 (Figure 4-7). The percentage of gathering collectors was higher in the spring of 2004 than 1974 and 1975 in four of the seven sites sampled in all years. Lower Miller, Tom McDonald and Lost Man below North Fork Creeks had lower spring percentages in 1974 and 1975 than 2004. In the late summer sampling period, gathering-collectors increased 155% in Elam Creek from 1974 to 2004 (Figure 4-8). Elam Creek, a site with no restoration, had the highest percentage of gathering collectors in the late summer of 2004 (79 percent), while South Fork Lost Man Creek, a site with recent disturbance, had the lowest (27 percent).

Scrapers graze upon periphyton in aquatic systems and their response to disturbance is variable (Harrington and Born, 2000). The percentage of scrapers was higher in 1974 than in 2004 at five of the eight sites sampled both years (Figure 4-9). In Bridge Creek, the percentage of spring scrapers was 12 times higher in 2004 than in 1974 and 1975. Spring percentage of scrapers reached above 30 percent in Hayes Creek and Lost Man Creek below North Fork in 1975. In 1975, spring percentage of scrapers increased 30-fold from 1974 in Hayes Creek. In five of the nine sites sampled both years,

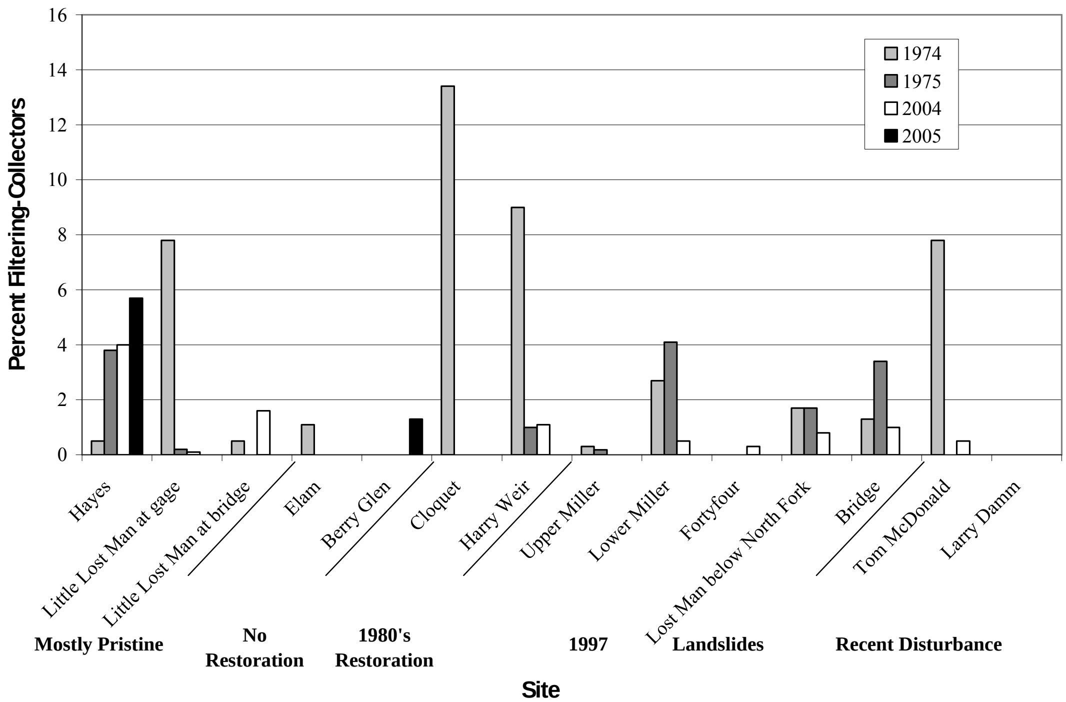


Figure 4-5. Spring percentage of filtering-collector macroinvertebrates sampled from tributaries of Redwood Creek in 1974, 1975, 2004 or 2005 with a 250μm Surber sampler.

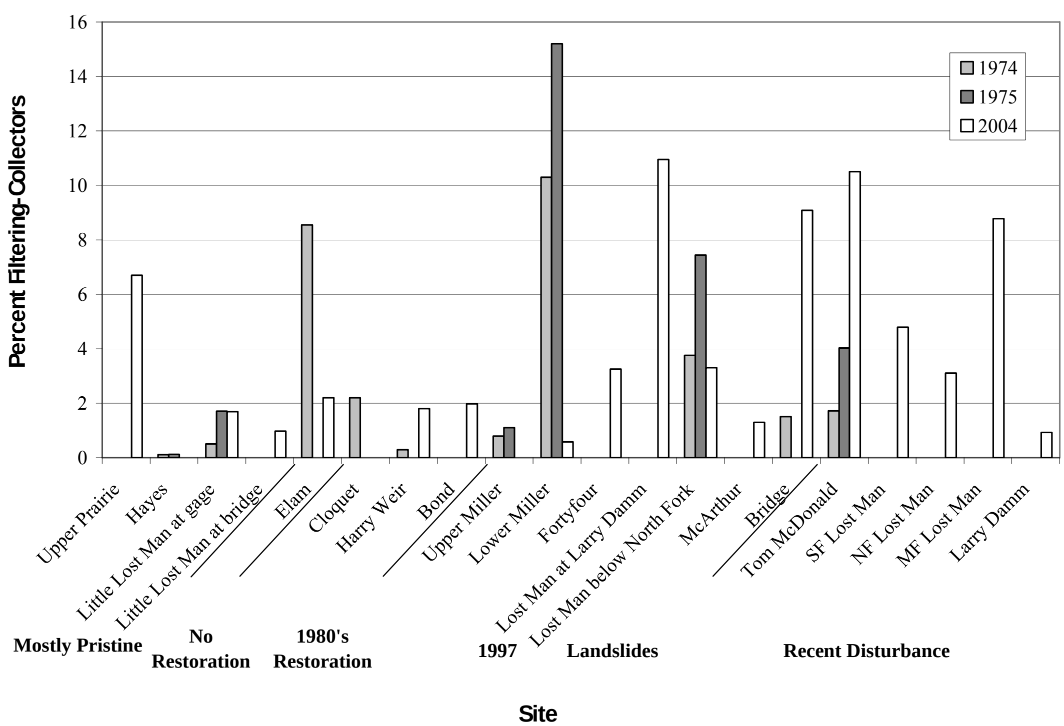


Figure 4-6. Summer percentage of filtering-collector macroinvertebrates sampled from tributaries of Redwood Creek in 1974, 1975 or 2004 with a 250μm Surber sampler.

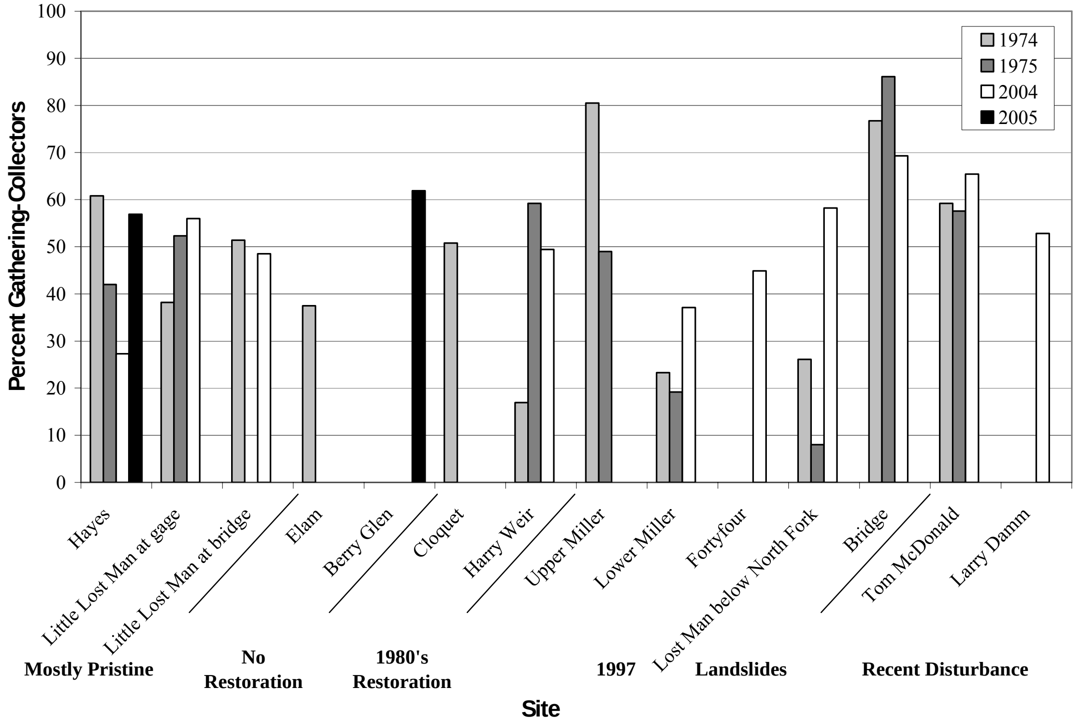


Figure 4-7. Spring percentage of gathering-collector macroinvertebrates sampled from tributaries of Redwood Creek in 1974, 1975, 2004 or 2005 with a 250μm Surber sampler.

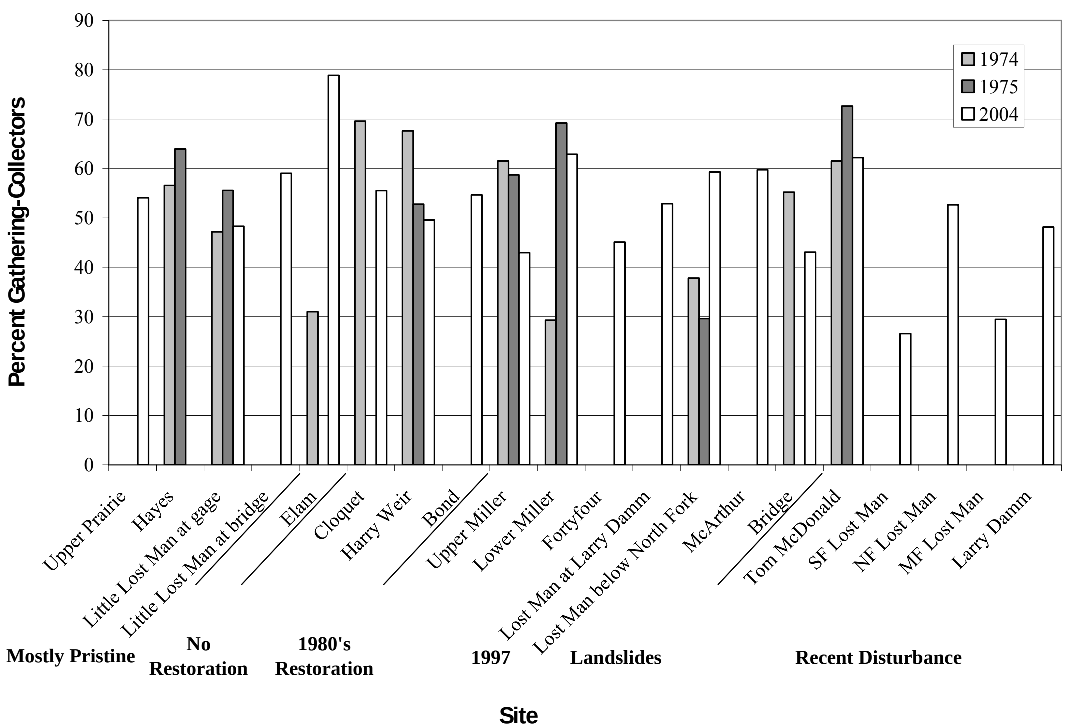


Figure 4-8 . Summer percentage of gathering-collector macroinvertebrates sampled from tributaries of Redwood Creek in 1974, 1975, or 2004 with a 250μm Surber sampler**.**

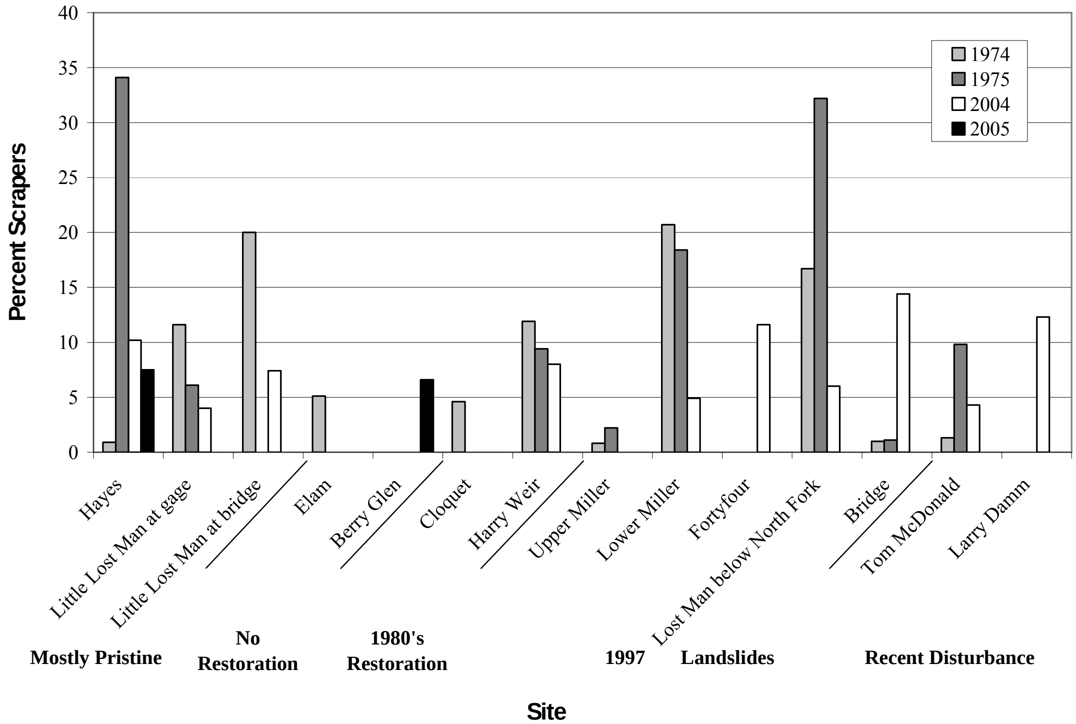


Figure 4-9. Spring percentage of scraper macroinvertebrates sampled from tributaries of Redwood Creek in 1974, 1975, 2004 or 2005 with a 250µm Surber sampler.

the percentage of late summer scrapers was higher in 2004 than in 1974 (Figure 4-10). Percentages of late summer scrapers were high in 2004 in four of the five recently disturbed sites as compared to other sites. Percentages of late summer scrapers were much higher in 1975 in Bridge Creek than in any other tributaries over all years. The channel of Bridge Creek aggraded several feet during the flood of 1975, and there was minimal riparian canopy at that point. There does not appear to be a consistent trend in the change in percentage of scrapers through time in either spring or summer.

Predators feed on other organisms and their response to disturbance is variable (Harrington and Born 2000). The spring percentage of predators was greater in 1974 than 2004 in five of the eight sites sampled both years (Figure 4-11). Predators in Hayes Creek, a pristine site, increased 150% from the spring of 1974 to 2004. Hayes Creek had a higher percentage of scrapers in the spring of 2005 than 1975. Late summer percentages of predators were higher in 2004 than 1974 in five of the nine tributaries sampled both years (Figure 4-12). Bridge Creek had a much higher percentage of predators in late summer 1975 than any other site over all years of sampling. There did not appear to be a trend between disturbance levels and percentage of predators.

Shredders make up the percentage of the macrobenthos that shreds coarse particulate matter and they are expected to decrease in response to disturbance (Harrington and Born 2000). Hayes and Tom McDonald Creeks were the only sites to have a higher percentage of shredders in the spring of 1974 than in 2004 (Figure 4-13). The spring percentage of shredders stayed low over all years at Lost Man Creek below North Fork and Bridge Creek, both disturbed in 1997. Upper Miller had a much higher percentage of shredders in the spring of 1975 than at any other time throughout the study, more than a three-fold increase from 1974. Percentages of late summer shredders were higher in 1974 than 2004 at five of the nine sites sampled both years (Figure 4-14). Elam Creek had a much higher percentage of late summer shredders in 1974 than did any other site during any year.

Piercer-herbivores made up less than 1 percent of the FFG composition in the 1970's samples with the exception of Bridge Creek in the late summer of 1975, where nine percent of the sample consisted of piercer-herbivores. There were no piercer-herbivores found in the 2004-2005 samples. Hydroptilidae were the only piercer-herbivores found in Surber samples collected in the 1970's. Omnivores were very rare in the 1970's samples (always less than 1 percent) and only present in two of the 2004-2005 samples. The only omnivore present was Turbellaria, which was in Berry Glen (2 percent) and Hayes (1 percent) Creeks in the spring of 2005. There may have been more Turbellarians present in the other 2004-2005 sites, but due to an identification error that was not corrected until the samples were finished they were not reported.

Invertebrates with Life Cycles of 2+ Years

The duration of aquatic insect life cycles can range from several weeks to several years. Merritt and Cummins (1996) term insect taxa that complete their life cycle in two years semivoltine, and merovoltine for those that require more than two years. In order to compare insects with either semivoltine or merovoltine life cycles between the 1970's and 2004-2005, the percentage of Odonates, Megalopterans, Perlids and Lara were

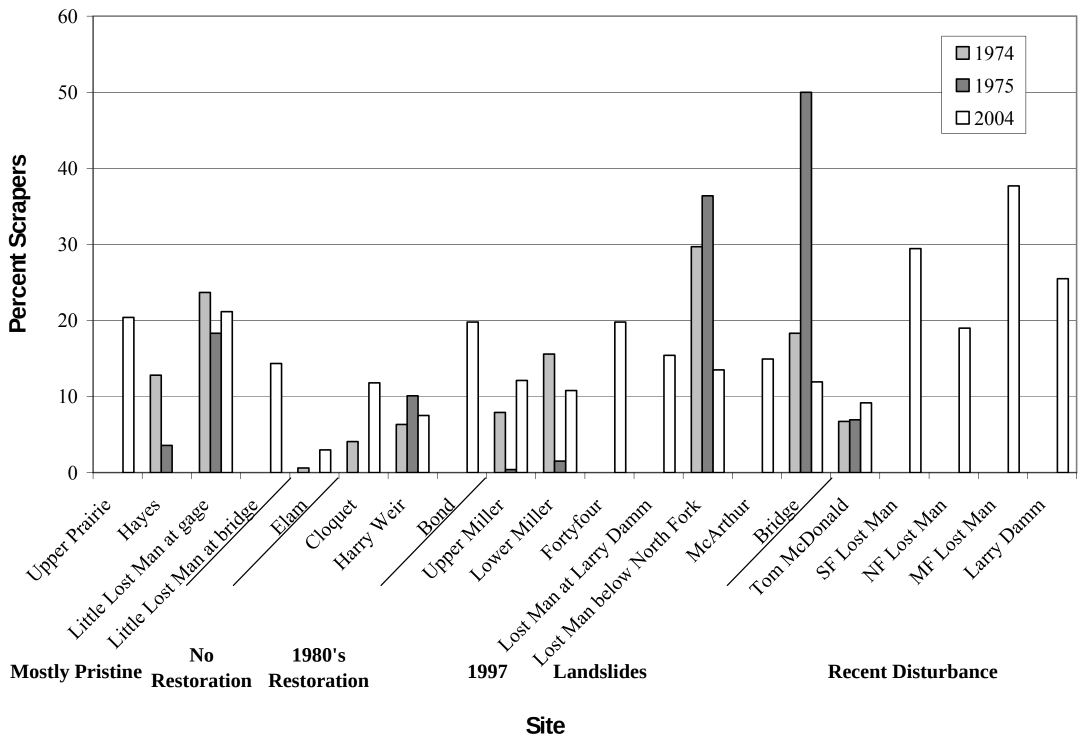


Figure 4-10. Summer percentage of scraper macroinvertebrates sampled from tributaries of Redwood Creek in 1974, 1975 or 2004 with a 250µm Surber sampler.

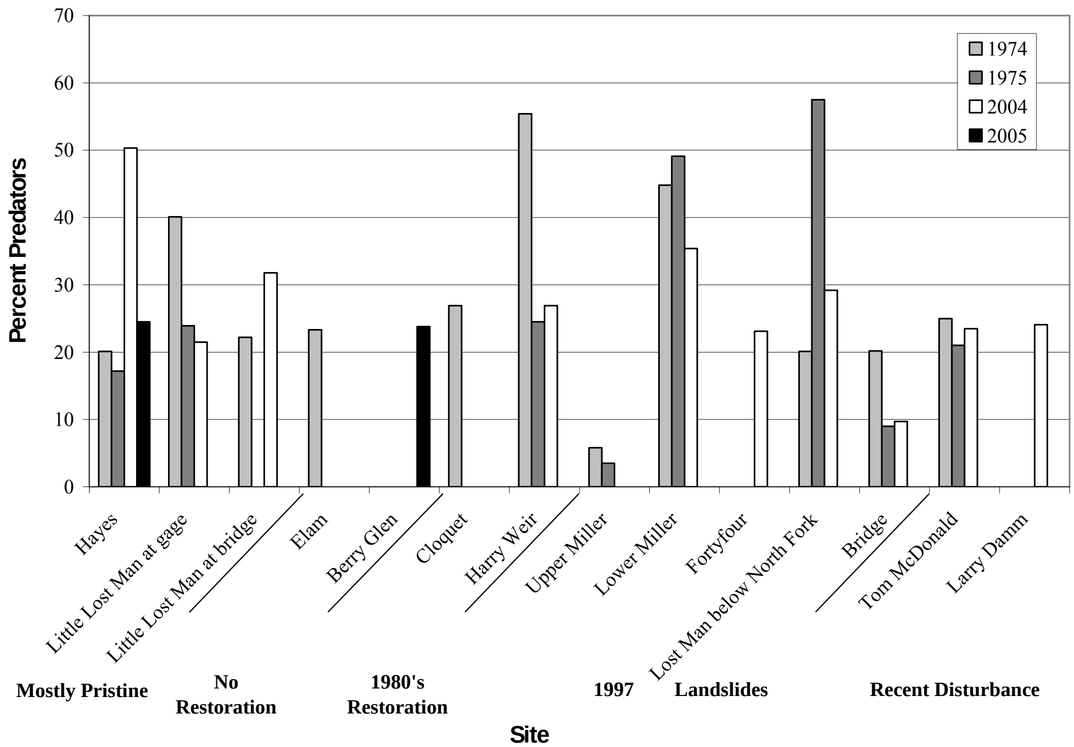


Figure 4-11. Spring percentage of predator macroinvertebrates sampled from tributaries of Redwood Creek in 1974, 1975, 2004 or 2005 with a 250µm Surber sampler.

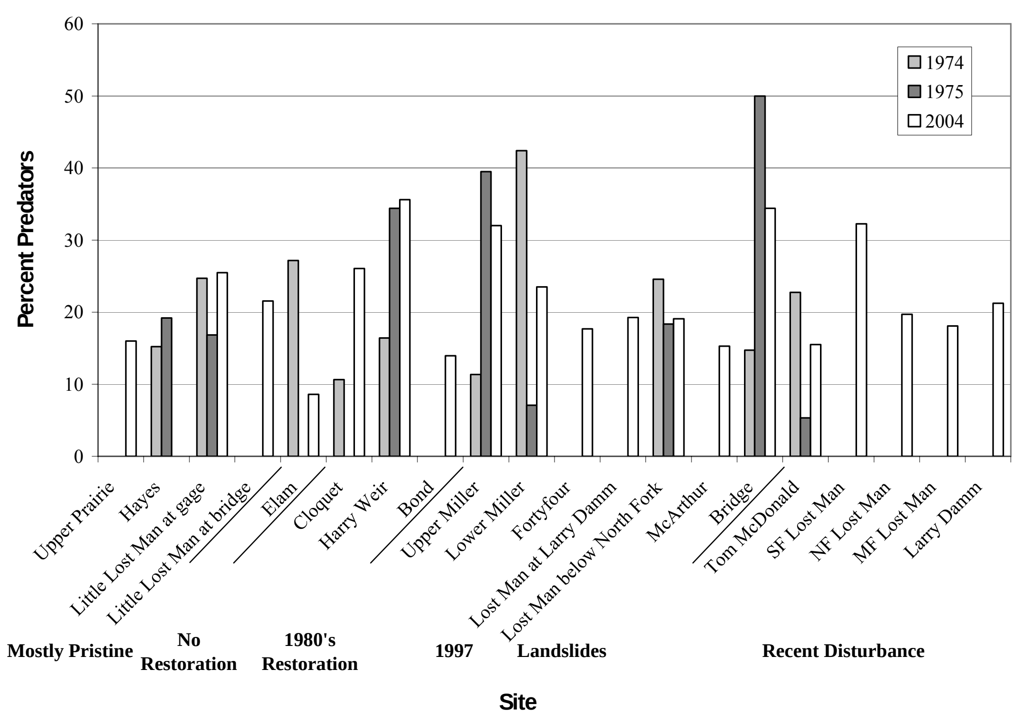


Figure 4-12. Summer percentage of predator macroinvertebrates sampled from tributaries of Redwood Creek in 1974, 1975 or 2004 with a 250µm Surber sampler.

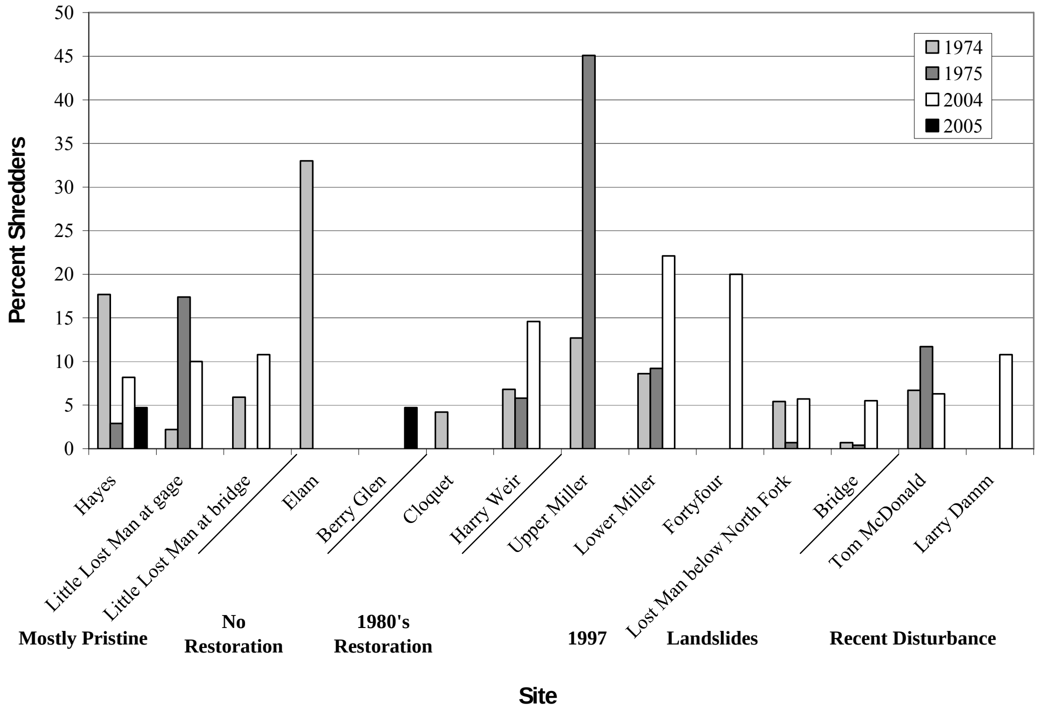


Figure 4-13. Spring percentage of shredder macroinvertebrates sampled from tributaries of Redwood Creek in 1974, 1975, 2004 or 2005 with a 250μm Surber sampler**.**

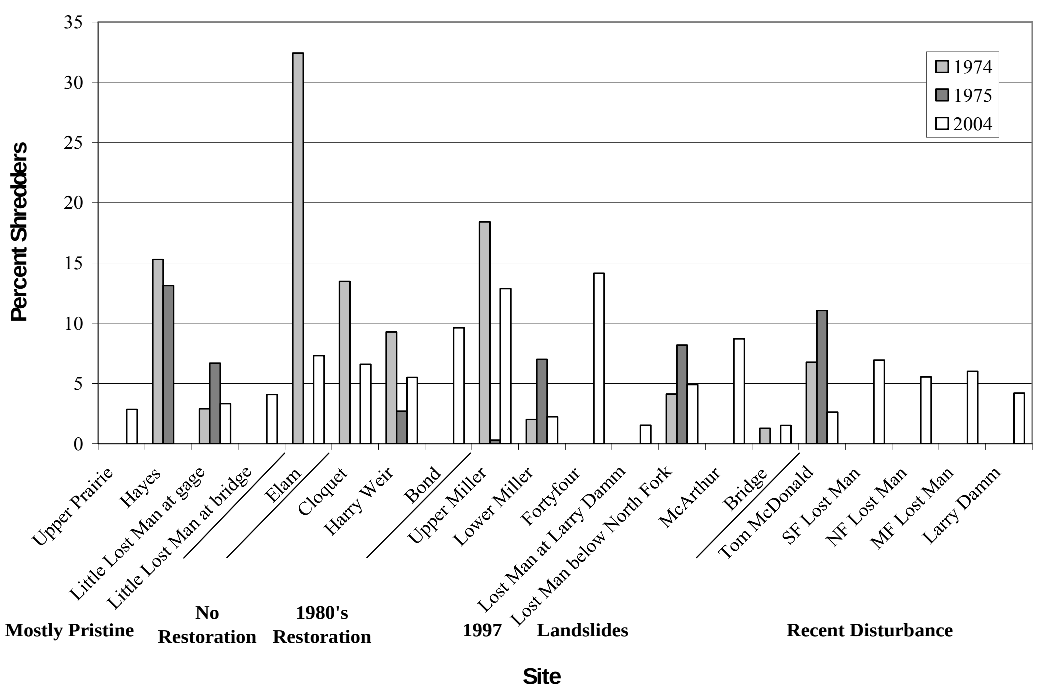


Figure 4-14. Summer percentage of shredder macroinvertebrates sampled from tributaries of Redwood Creek in 1974, 1975 or 2004 with a250μm Surber sampler**.**

calculated. All of these insects which occur in the study area require two or more years to grow from egg to adult. Percentages of insects exhibiting 2+ year life cycles were higher in every site in the spring of 2004 than 1974 and were highest in Fortyfour Creek (11 percent) in 2004 (Fig. 4-15). Hayes Creek, a pristine site, had 0 percent long-lived insects, but this site goes dry in the summer. Bridge Creek, with a very mobile channel bed, had 0 percent insects with extended life cycles in the spring of 1975 and 2005. In the spring of 1975, Lower Miller Creek had the highest percentage of insects with extended life cycles. Percentages of taxa with longer life cycles were higher in the late summer of 2004 than 1974 in seven of the nine sites sampled in both years (Figure 4-16). Harry Weir Creek had the highest percentage of insects with 2+ year life cycles in 2004 (5.5 percent), five times greater than that in both 1974 and 1975. Lower Miller Creek had the highest percentage of insect taxa with 2+ year life cycles in the late summer of 1975, while Bridge Creek had 0 percent.

Benthic Kick Samples

Taxa richness is the total number of distinct groups or taxa in a sample and reflects the diversity of the aquatic assemblage. Taxa richness is expected to decrease in response to disturbance (Harrington and Born, 2000). The benthic invertebrate community consisted of 181 taxa in the 2004 and 2005 sampling season. Benthic kick samples were collected from sites in the summer of 2004 and 2005 and in the spring of 2005. In the spring of 2005, total taxa richness was highest in Harry Weir Creek and lowest in Bond Creek, which both had road restoration in the 1980's (Figure 4-17). Total taxa richness was higher in the summer of 2005 than 2004 in 14 of the 19 sites sampled in both years (Figure 4-18). Taxa richness was highest in the summer of 2005 in Upper Miller Creek, which underwent restoration in the 1980's, and lowest in Elam and Fortyfour Creeks. There did not appear to be any consistent trend between level of disturbance and taxa richness in either year.

EPT taxa richness is the number of taxa in the Ephemeroptera (mayfly), Plecoptera (stonefly) and Trichoptera (caddisfly) orders and is expected to decrease in response to disturbance (Harrington and Born, 2000). These three orders are considered sensitive to pollution and act as indicator species to determine the level of human disturbance and impact in an aquatic system. Spring EPT taxa richness was highest in South Fork Lost Man Creek and lowest in Bridge Creek (Figure 4-19). EPT taxa richness was higher in the summer of 2005 than in the summer of 2004 in 11 of the 19 sites sampled in both years (Figure 4-20). EPT taxa richness was the highest in the summer of 2004 at South Fork Lost Man Creek and was lowest in the summer of 2005 at Elam Creek. There did not appear to be any consistent trend between level of disturbance and EPT taxa richness.

Ephemeroptera taxa richness is the number of mayfly taxa in a sample and is expected to decrease in response to disturbance (Harrington and Born, 2000). Spring Ephemeroptera taxa richness was highest in South Fork Lost Man Creek and lowest in Hayes Creek (Figure 4-21). Summer Ephemeroptera taxa richness was the same or higher in 2005 than 2004 in 18 of the 19 sites sampled in both years (Figure 4-22). Elam Creek was the only site in which numbers of Ephemeroptera taxa were larger in the summer of 2004 than 2005. Ephemeroptera taxa richness was three times greater in the summer of 2005 than 2004 in Bridge Creek. Ephemeroptera richness was highest in

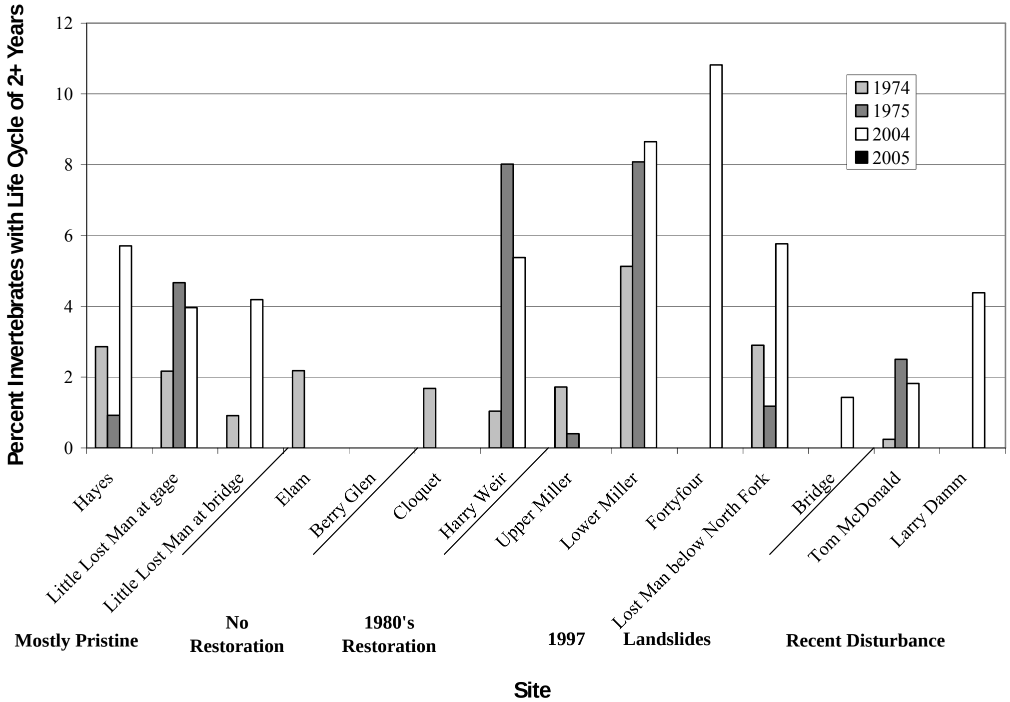


Figure 4-15. Spring percentage of invertebrates with life cycles of two years or more sampled from tributaries of Redwood Creek in 1974, 1975, 2004 or 2005 with a 250μm Surber sampler.

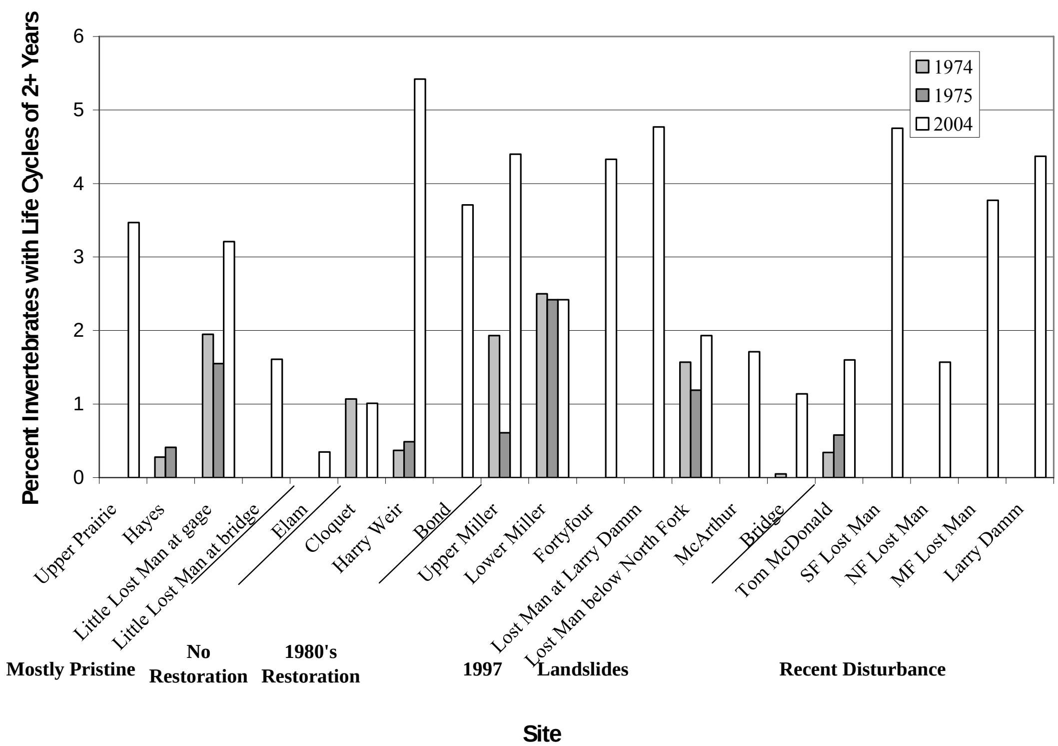


Figure 4-16. Summer percentage of invertebrates with life cycles of two years or more sampled from tributaries of Redwood Creek in 1974, 1975 or 2004 with a Surber sampler.

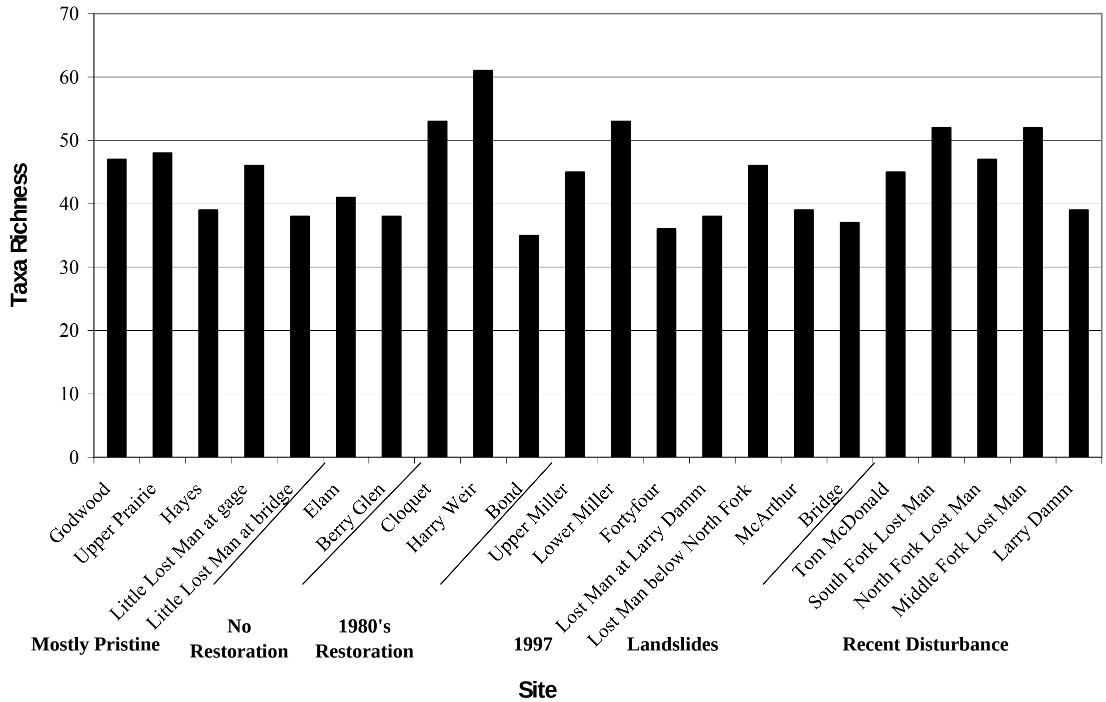


Figure 4-17. Spring macroinvertebrate taxa richness sampled from tributaries of Redwood Creek in 2005 with a 500µm benthic kick net.

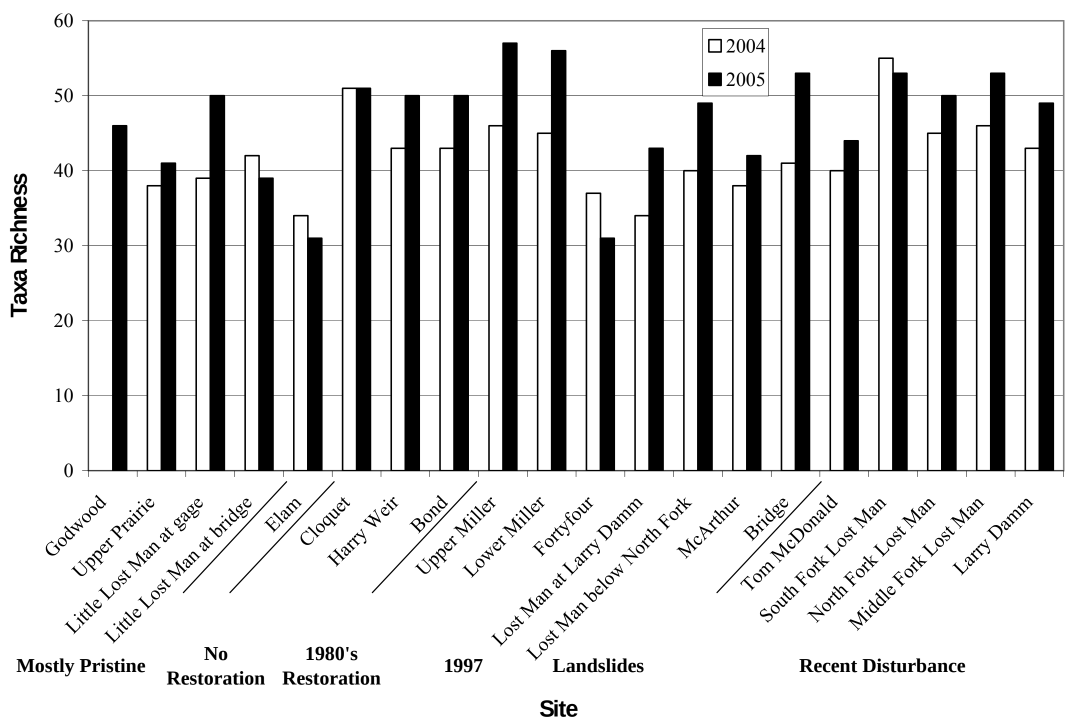


Figure 4-18. Summer macroinvertebrate taxa richness sampled from tributaries of Redwood Creek in 2004 or 2005 with a 500µm benthic kick net.

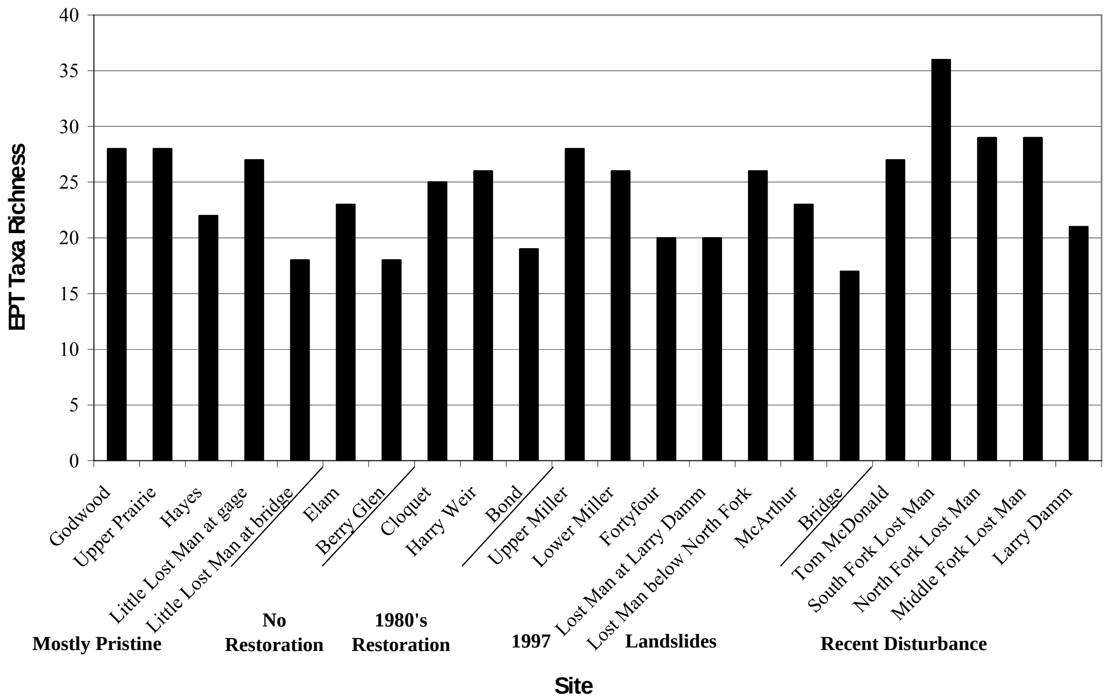


Figure 4-19. Spring EPT taxa richness sampled from tributaries of Redwood Creek in 2005 with a 500µm benthic kick net.

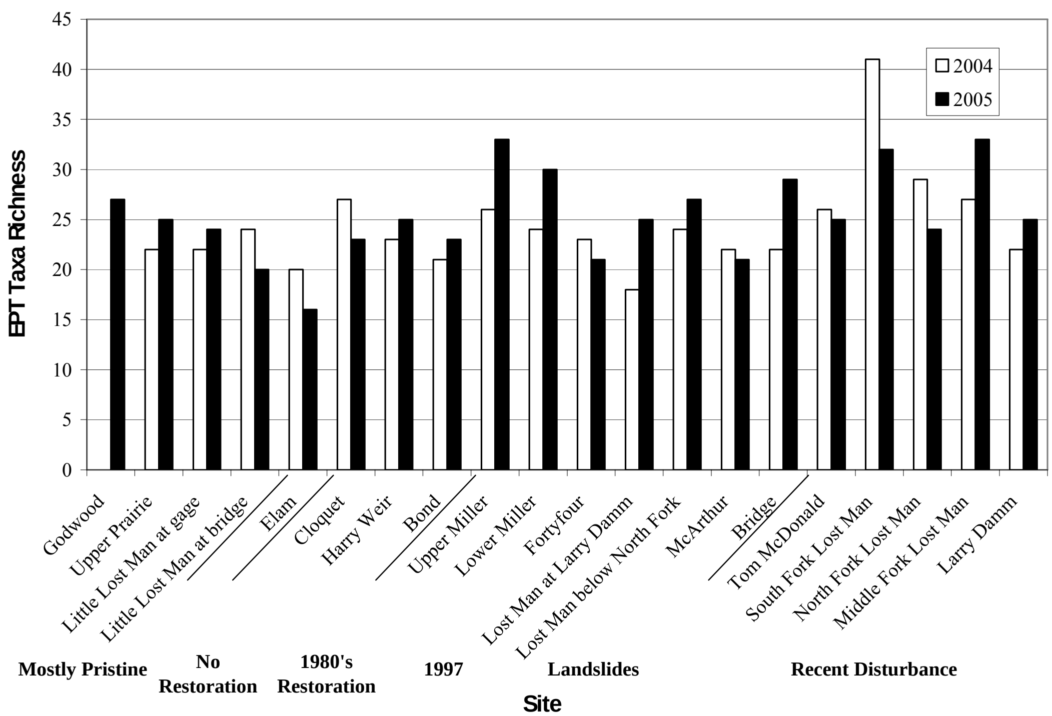


Figure 4-20. Summer EPT taxa richness sampled from tributaries of Redwood Creek in 2004 or 2005 with a 500µm benthic kick net.

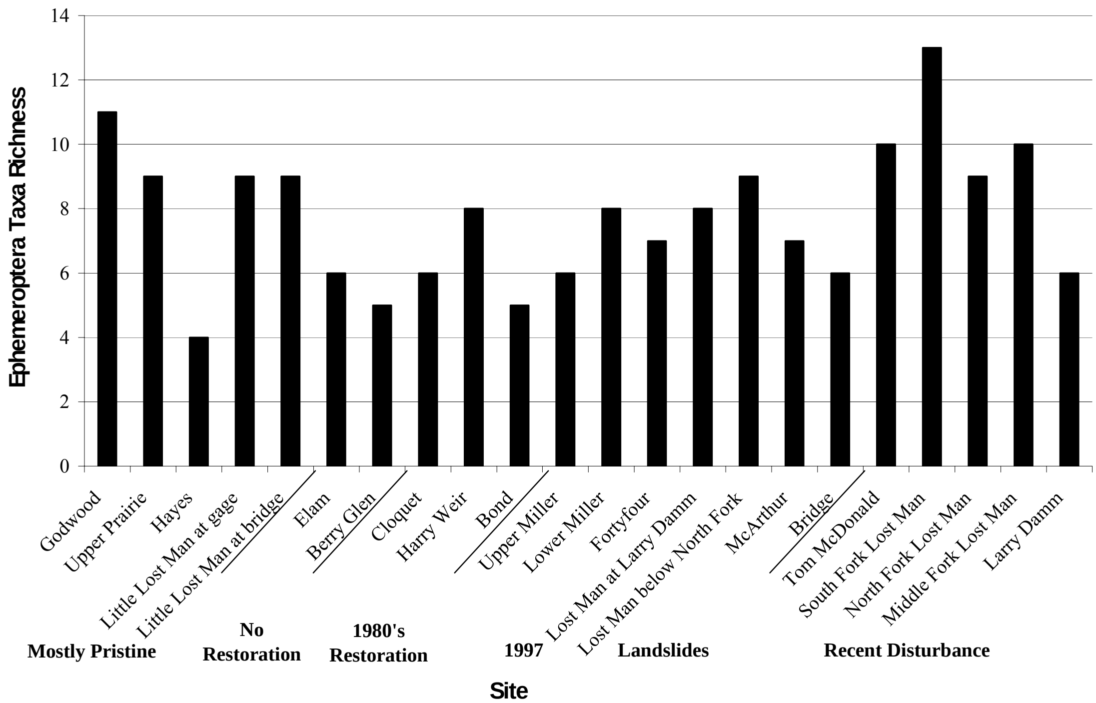


Figure 4-21. Spring Ephemeroptera taxa richness sampled from tributaries of Redwood Creek in 2005 with a 500µm benthic kick net.

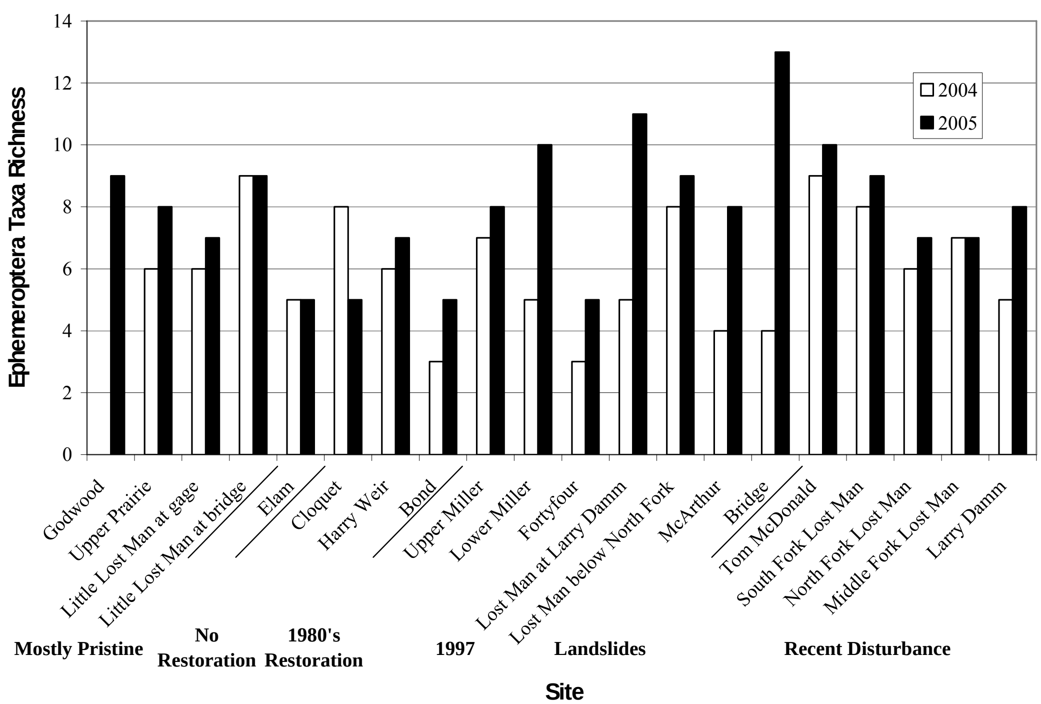


Figure 4-22. Summer Ephemeroptera taxa richness sampled from tributaries of Redwood Creek in 2004 or 2005 with a 500µm benthic kick net.

Bridge Creek in the summer of 2005 and lowest in Bond and Fortyfour Creeks in the summer of 2004.

Plecoptera taxa richness is the number of stonefly taxa in a sample and is expected to decrease in response to disturbance (Harrington and Born, 2000). Plecoptera taxa disappear as riparian vegetation is lost and sediment clogs the interstitital spaces among cobbles (Karr and Chu, 1999). Spring Plecoptera taxa richness was highest in South Fork Lost Man Creek and lowest in Bridge Creek (Figure 4-23). Spring Plecoptera taxa richness stayed at or above nine in all five recently disturbed sites. Plecoptera taxa richness was the same or lower in 2005 than in 2004 in 15 of the 19 sites sampled in both years (Figure 4-24). Plecoptera taxa richness was the highest in Middle Fork Lost Man Creek in the summer of 2005 and lowest in Lost Man Creek below North Fork and Little Lost Man Creek at the bridge in the summer of 2004 and 2005 respectively. There did not appear to be any consistent trend between level of disturbance and Plecoptera taxa richness.

Trichoptera taxa richness is the number of caddisfly taxa in a sample and is expected to decrease in response to disturbance (Harrington and Born, 2000). Spring Trichoptera taxa richness was greatest at Upper Prairie Creek and smallest at Little Lost Man Creek at the bridge (Figure 4-25). In four of the five pristine sites, Trichoptera richness stayed at or above 9. Trichoptera taxa richness was the same or higher in the summer of 2005 than 2004 in 12 of the 19 sites sampled both years (Figure 4-26). Trichoptera taxa richness was greatest in the summer of 2004 in South Fork Lost Man Creek and lowest in the summer of 2005 in Elam Creek.

The EPT index is the percentage composition of mayfly, stonefly and caddisfly larvae present in the stream and is expected to decrease in response to disturbance (Harrington and Born, 2000). Metrics that measure percentage composition reflect the relative contribution of a population of particular taxa to the total fauna. The spring EPT index was over 70 percent in six of the 22 sites sampled (Figure 4-27). The EPT index dropped to 30 percent in Berry Glen Creek in the spring. The summer EPT index was lower in 2005 than 2004 in 15 of the 19 sites sampled in both years (Figure 4-28). The EPT index dropped to below 20 percent in Elam and Fortyfour Creeks in the summer of 2005 and in Elam Creek, the EPT index remained under 30 percent in the summer of 2004. The summer EPT index decreased by half from 2004 to 2005 in Fortyfour Creek. Between the spring and the summer of 2005, the EPT index at Elam, Bond and Fortyfour Creeks decreased by at least 50 percent.

The sensitive EPT index is the percentage composition of mayfly, stonefly and caddisfly larvae with a tolerance value of 0 through 3 and is expected to decrease in response to disturbance (Harrington and Born, 2000). The spring sensitive EPT index was highest at Upper Miller Creek and lowest at Bridge Creek (Figure 4-29). The summer sensitive EPT index was lower in 2005 than in 2004 in 13 of the 19 sites sampled in both years (Figure 4-30). The summer sensitive EPT index dropped by more than 50 percent between the summers of 2004 and 2005 in Elam and Fortyfour Creeks. Between the spring and summer of 2005, the sensitive EPT index at Fortyfour, Bridge and Elam Creeks decreased by at least half.

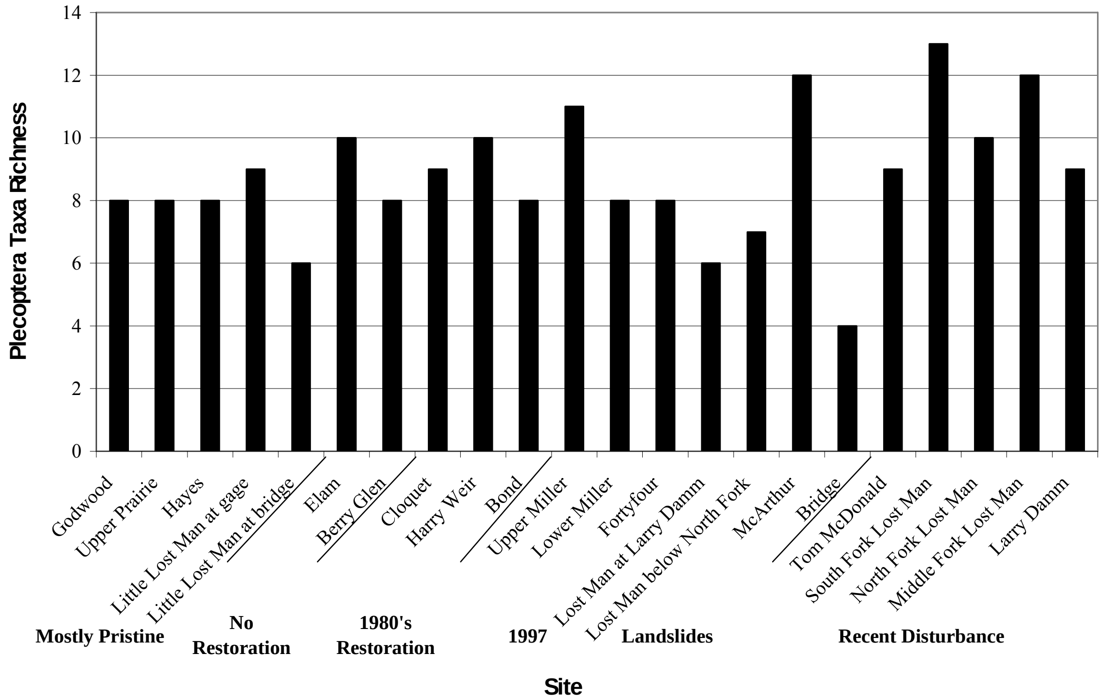


Figure 4-23. Spring Plecoptera taxa richness sampled from tributaries of Redwood Creek in 2005 with a 500µm benthic kick net.

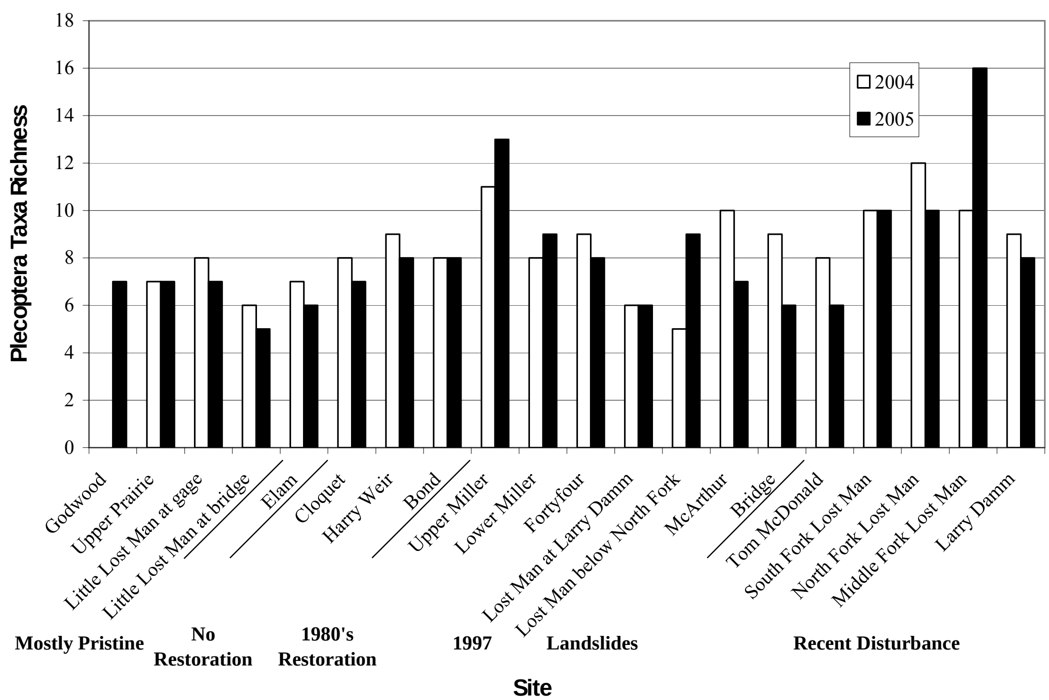


Figure 4-24. Summer Plecoptera taxa richness sampled from tributaries of Redwood Creek in 2004 or 2005 with a 500µm benthic kick net.

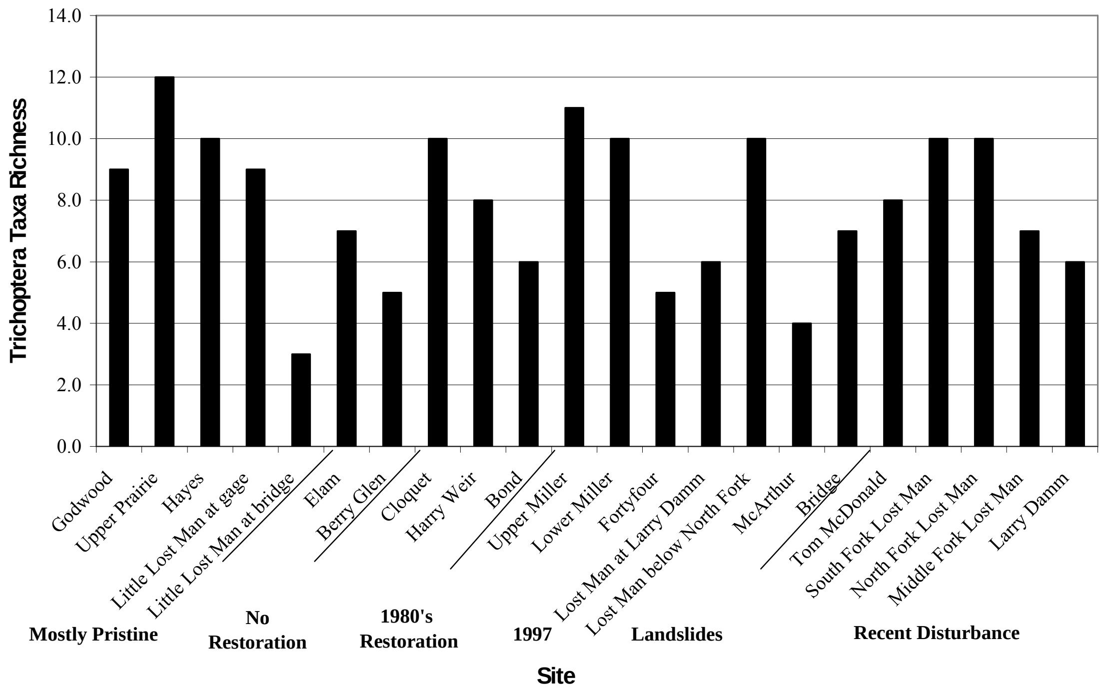


Figure 4-25. Spring Trichoptera taxa richness sampled from tributaries of Redwood Creek in 2005 with a 500µm benthic kick net.

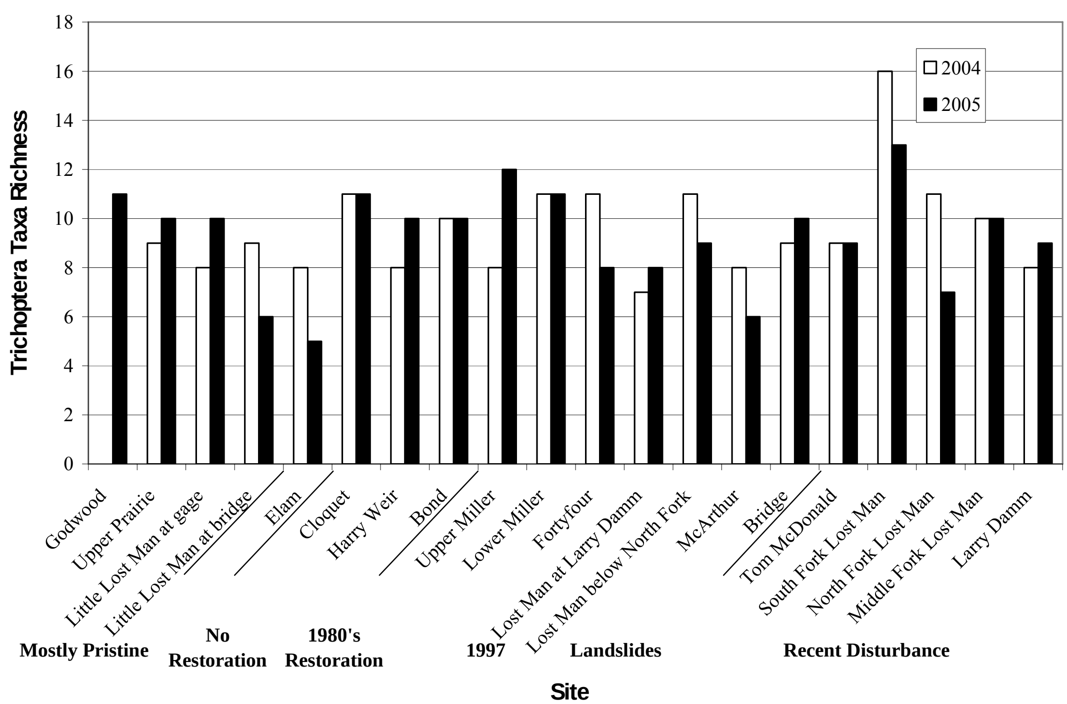


Figure 4-26. Summer Trichoptera taxa richness sampled from tributaries of Redwood Creek in 2004 or 2005 with a 500µm benthic kick net.

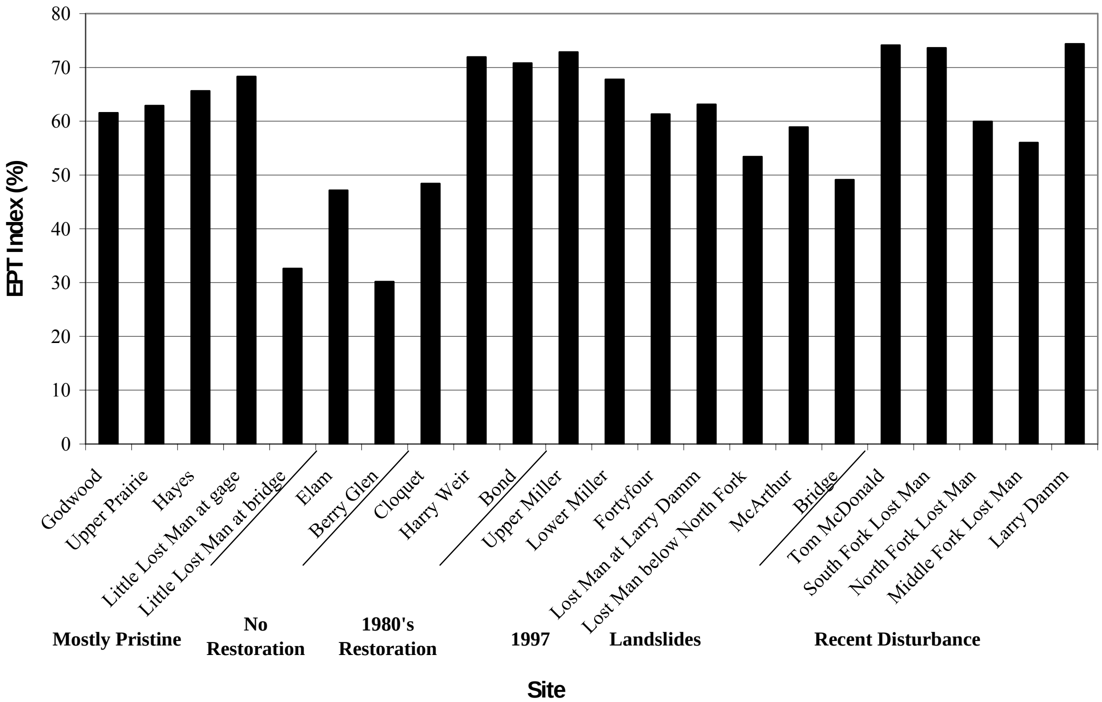


Figure 4-27. Spring EPT index sampled from tributaries of Redwood Creek in 2005 with a 500µm benthic kick net.

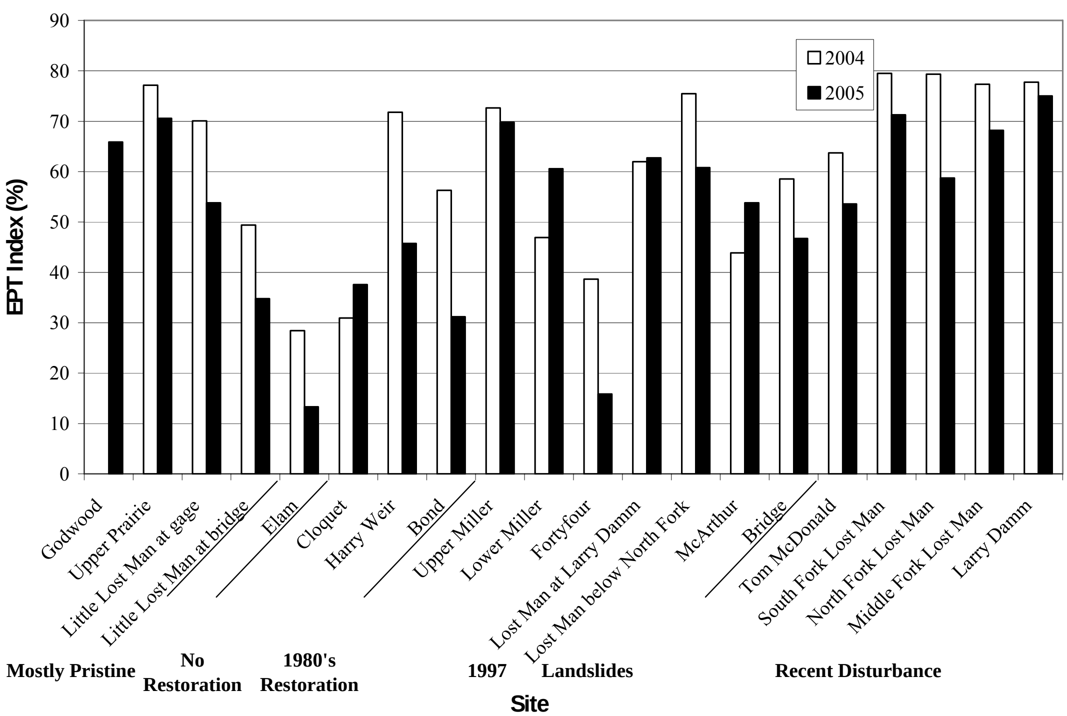


Figure 4-28. Summer EPT index sampled from tributaries of Redwood Creek in the summer of 2004 or 2005 with a 500µm benthic kick net.

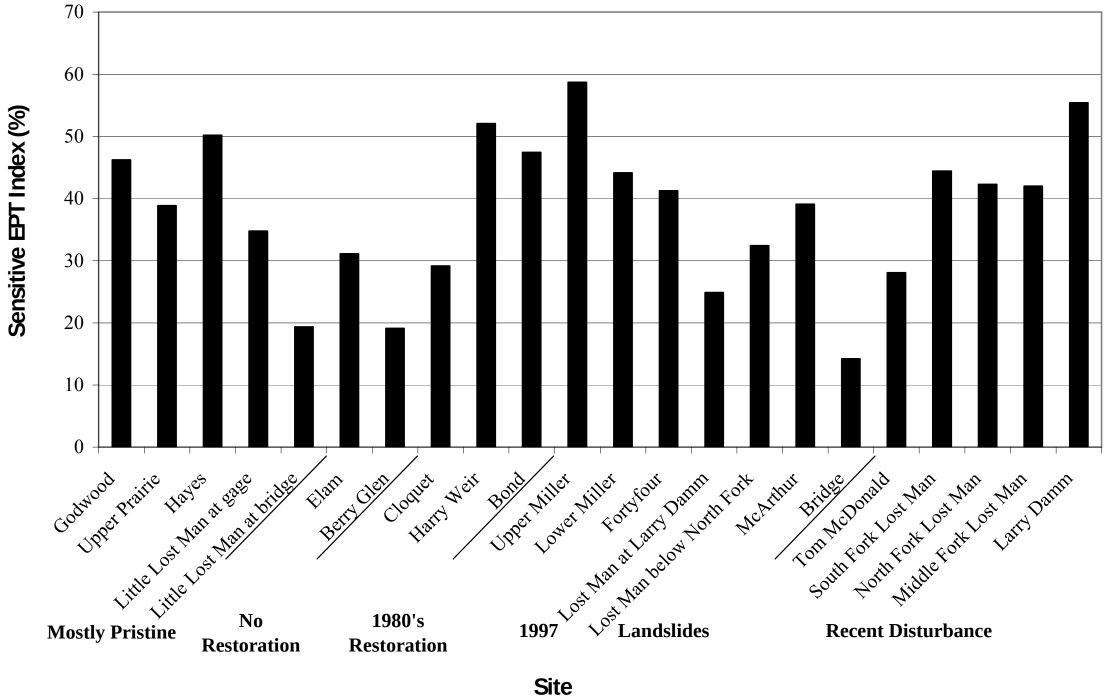


Figure 4-29. Spring sensitive EPT index sampled from tributaries of Redwood Creek in 2005 with a 500µm benthic kick net.

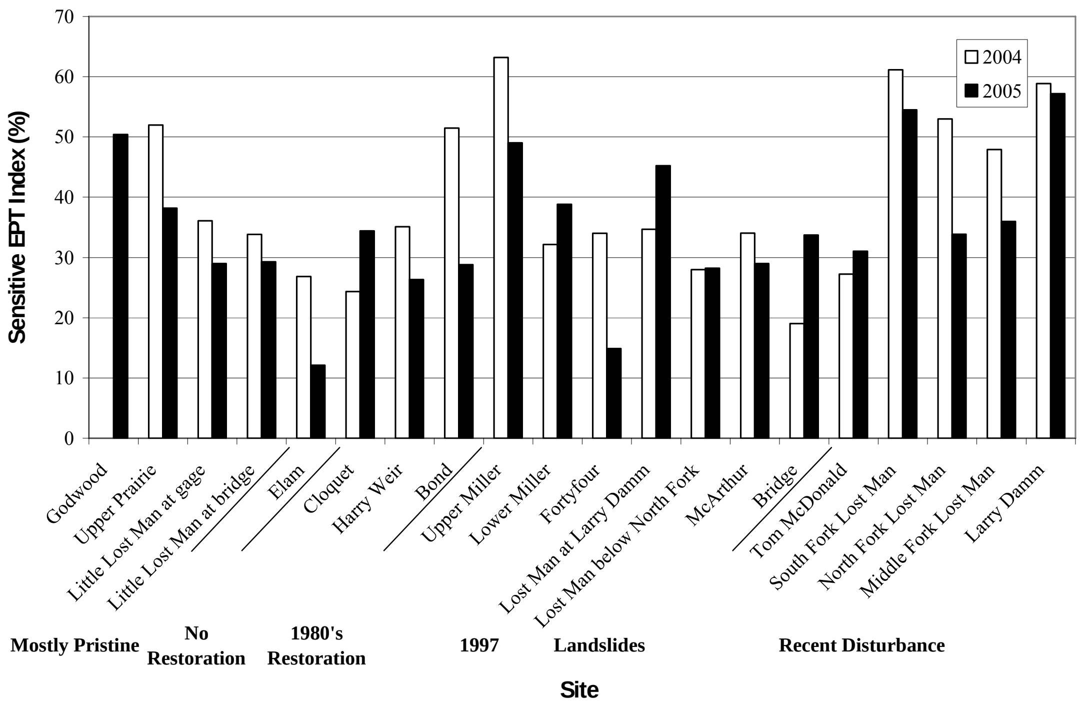


Figure 4-30. Summer sensitive EPT index sampled from tributaries of Redwood Creek in 2004 or 2005 with a 500µm benthic kick net.

The percentage of Baetidae in a sample is the percentage composition of mayflies that are in the more tolerant family Baetidae. The percentage of Baetids is expected to increase in response to disturbance, notably in the form of fine particulate organic matter and sedimentation (Harrington and Born, 2000). The percentage of Baetids fluctuated greatly among sites in both the spring and summer samples. The spring percentage of Baetidae was highest in Tom McDonald Creek, a recently disturbed site, and lowest in Middle Fork Lost Man Creek, disturbed around 1997, a 20-fold difference (Figure 4-31). The summer percentage of Baetidae was less in 2004 than in 2005 in 11 of the 19 sites sampled in both years (Figure 4-32). The differences between the summers of 2004 and 2005 were generally quite large. There was less than 1 percent Baetids in both the summer of 2004 and 2005 in Elam Creek, which has not undergone any restoration. The percentage of Baetidae was greater in the spring than the summer of 2005 in 14 of the 20 sites sampled in both seasons.

The percentage of Hydropsychidae in a sample is the percentage composition of caddisflies that are in the more tolerant family Hydropsychidae. As with the Baetids, the percentage of Hydropsychids is expected to increase in response to disturbance, notably in the form of fine particulate organic matter and sedimentation (Harrington and Born, 1999). Percentage of Baetids and percentage of Hydropsychids are regional metrics that have evolved to be useful in California (Harrington and Born, 2000). The percentage of Hydropsychids fluctuated greatly among sites in both the spring and summer samples. The percentage of Hydropsychids was generally quite low (<1.5 percent) in the spring with the exception of Godwood Creek (4 percent) (Figure 4-33). There were no Hydropsychids present in the spring in Little Lost Man Creek at the bridge, Lost Man Creek upstream of Larry Damm Creek, Cloquet, Fortyfour and Larry Damm Creeks. The percentage of Hydropsychids remained at or below 1 percent in Elam, Upper Miller and Fortyfour Creeks in both the summer of 2004 and 2005 (Figure 4-34). The percentage of Hydropsychids was highest in the summer of 2005 in Middle Fork Lost Man Creek. The percentage of Hydropsychids in samples was generally larger in the summer of 2005 than in the spring.

The percentage composition of non-insecta taxa in samples is expected to increase in response to impairment (Harrington and Born, 2000). The percentage of non-insecta taxa in the spring was 60 percent in Berry Glen Creek, yet remained at 20 percent or below in the remaining sites in the spring (Figure 4-35). Berry Glen Creek was only sampled in the spring of 2005 and had a very high number of Amphipoda in the sample, leading to its high percentage of non-insecta taxa. The percentage of non-insecta taxa was higher in the summer of 2005 than 2004 in 14 of the 19 sites sampled in both years (Figure 4-36). It was highest in both summers in Cloquet Creek. The difference in the percentage of non-insecta taxa between the summer of 2004 and 2005 at least doubled in Upper Prairie, Lower Miller, Tom McDonald, North Fork Lost Man and Larry Damm Creeks.

In order to determine the sensitivity of an organism to disturbance, a tolerance value is assigned to each individual taxa. Each taxa is assigned a tolerance value of 0 (highly intolerant) to 10 (highly tolerant). The number of individuals of each taxa in a sample are multiplied by their tolerance value, added together and divided by the total number of organisms in the sample to obtain a final tolerance value. This metric is based on the Hilsenhoff Biotic Index that uses a set of taxon-specific tolerance values to

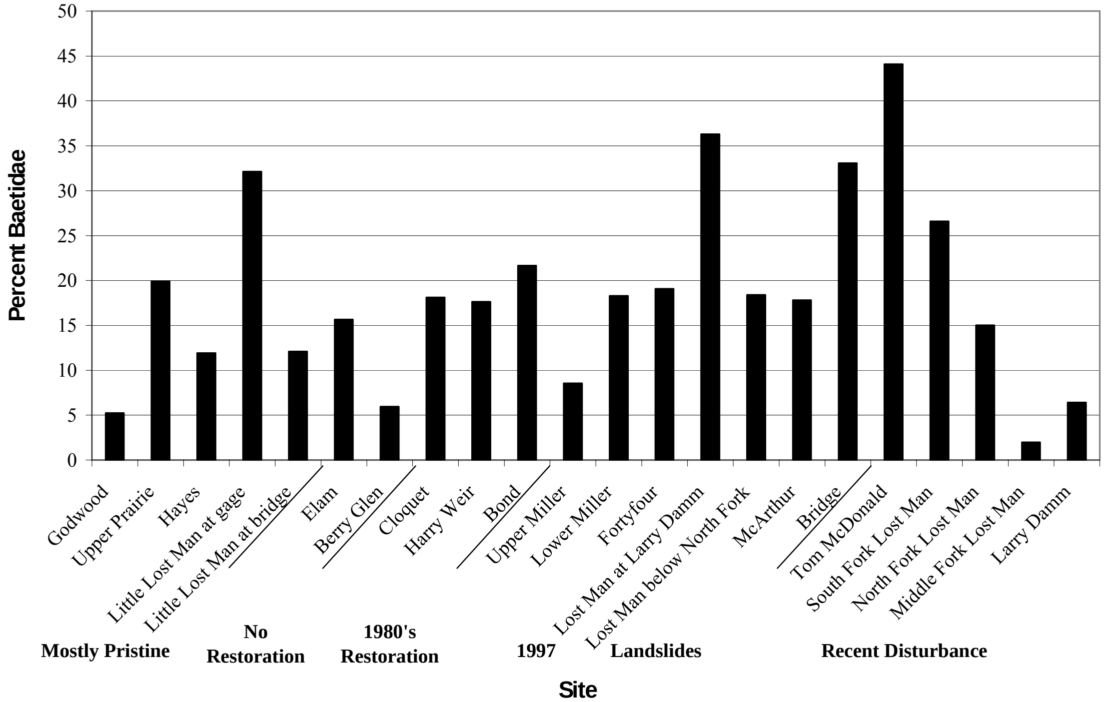


Figure 4-31. Spring percentage of Baetidae sampled from tributaries of Redwood Creek in 2005 with a 500µm benthic kick net.

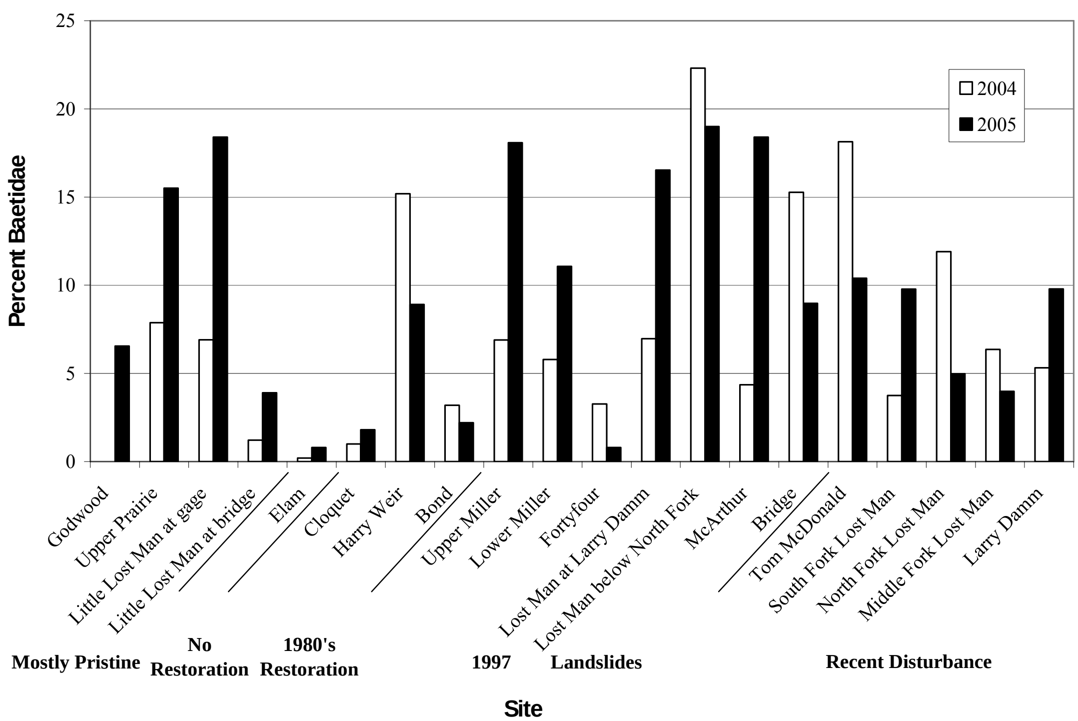


Figure 4-32. Summer percentage of Baetidae sampled from tributaries of Redwood Creek in 2004 or 2005 with a 500µm benthic kick net.

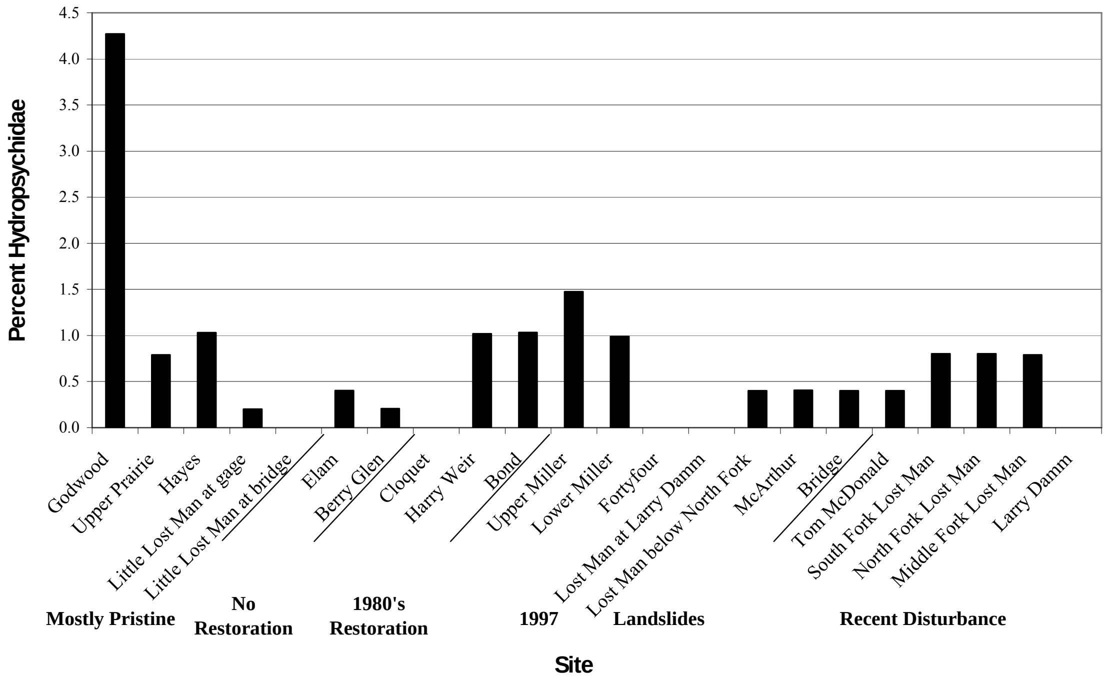


Figure 4-33. Spring percentage of Hydropsychidae sampled from tributaries of Redwood Creek in 2005 with a 500µm benthic kick net.

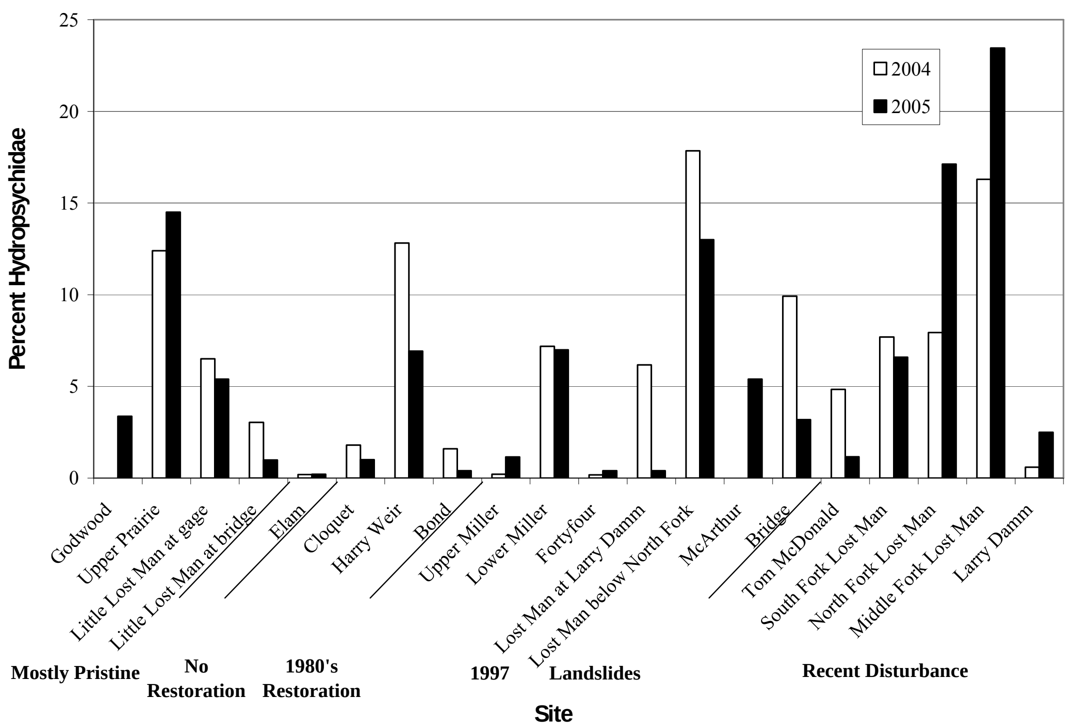


Figure 4-34. Summer percentage of Hydropsychidae sampled from tributaries of Redwood Creek in 2004 or 2005 with a 500µm benthic kick net.

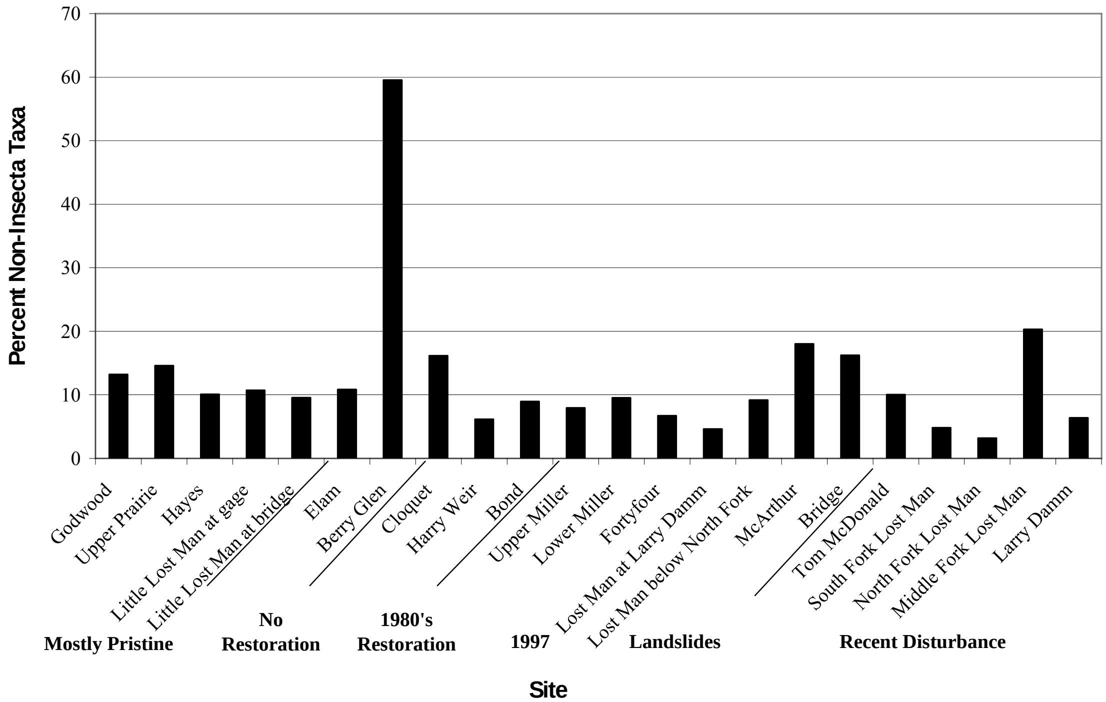


Figure 4-35. Spring percentage of non-insecta taxa sampled from tributaries of Redwood Creek in 2005 with a 500µm benthic kick net.

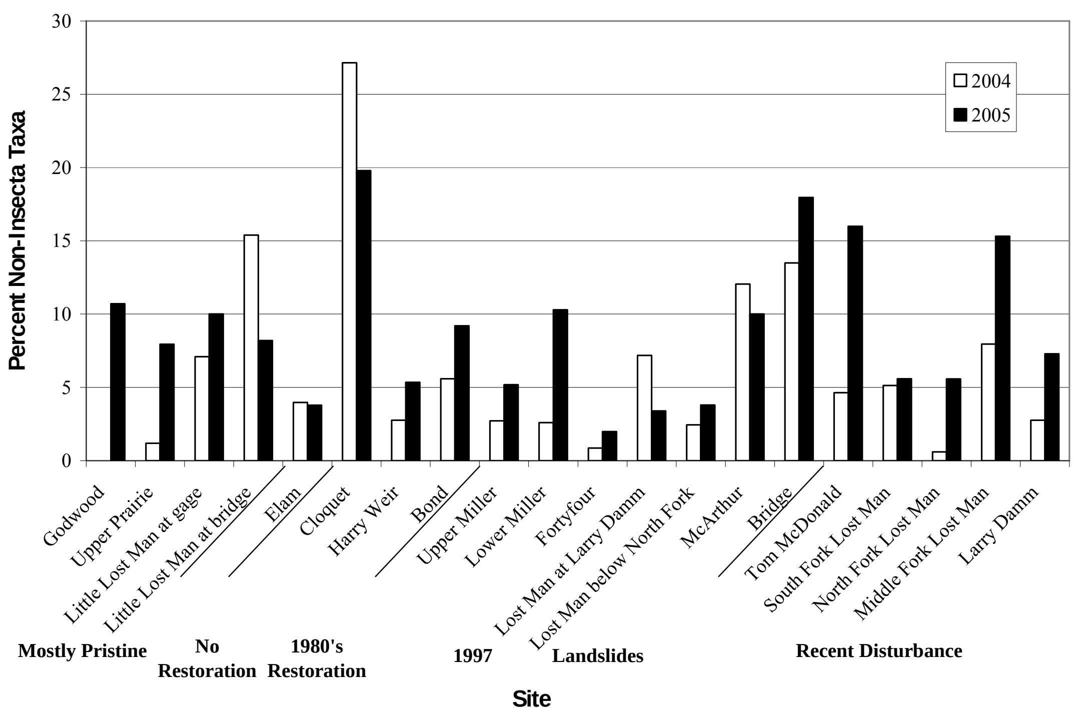


Figure 4-36. Summer percentage of non-insecta taxa sampled from tributaries of Redwood Creek in 2004 or 2005 with a 500µm benthic kick net.

calculate an overall, or community level, tolerance (Hilsenhoff, 1987). This metric was originally designed to serve as a measure of community tolerance to organic pollution in Wisconsin streams, but is commonly used as a general index of pollution tolerance (CAMLnet, 2003). The original values are regionally specific and different organisms can respond uniquely to different disturbance situations, so the application of this metric should be treated with caution. The tolerance values used in this study and by the California State Aquatic Bioassessment Laboratory were assigned by Aquatic Biology Associates, Inc. in Corvallis, Oregon, unless a taxon found in California is not assigned a value in the Pacific Northwest, in which case the EPA value for Idaho is used (CAMLnet, 2003).

The tolerance value is expected to increase in response to disturbance (Harrington and Born, 2000). The spring tolerance value was highest in Berry Glen Creek and lowest in Upper Miller Creek (Figure 4-37). Tolerance values in the spring ranged from 2.8 to 4.7. The tolerance value was higher in the summer of 2005 than 2004 in 16 of the 19 sites sampled in both years (Figure 4-38). The tolerance value at Lost Man Creek upstream of Larry Damm Creek, Cloquet Creek and Miller Creek was smaller in the summer of 2005 than 2004. Tolerance values in the summers also ranged from 2 to 5. There did not appear to be any trend between tolerance value and level of disturbance.

The percentage of intolerant organisms is the percentage of organisms in a sample that are highly intolerant to disturbance as indicated by a tolerance value of 0, 1 or 2 and is expected to decrease in response to disturbance (Harrington and Born, 2000). This metric has been corrected for watershed area. Harry Weir Creek, which had road restoration in the 1980's, had the highest spring percentage of intolerant organisms (47 percent) (Figure 4-39). Four of the 22 sites, Little Lost Man Creek at the bridge, Berry Glen Creek, Cloquet Creek and Bridge Creek, had less than 20 percent intolerant organisms in the spring. The percentage of intolerant organisms was higher in the summer of 2004 than 2005 in 13 of the 19 streams sampled in both the summers (Figure 4-40). Upper Miller Creek had the highest percentage of intolerant organisms in the summer of 2004 (61 percent). Larry Damm Creek had greater than 50 percent of intolerant organisms in both summers. Fortyfour and Elam Creeks had less than 10 percent of intolerant organisms in the summer of 2005.

The percentage of tolerant organisms is the percentage of organisms in a sample that are highly tolerant to disturbance as indicated by a tolerance value of eight, nine or ten and is expected to increase in response to disturbance (Harrington and Born, 2000). The percentage of tolerant organisms in the spring was low, always less than 2.5 percent (Figure 4-41). Cloquet Creek, restored in the 1980's, had the highest percentage of tolerant organisms in the spring. Little Lost Man Creek at the gage, Lost Man Creek downstream of North Fork, and Hayes, Bond, South Fork Lost Man, McArthur, Tom McDonald and North Fork Lost Man Creeks had no tolerant organisms present in the spring. All streams had tolerant organisms present in the summer of 2005, although only nine of the 19 sites sampled in the summer of 2004 had tolerant organisms present (Figure 4-42). Bridge Creek had the highest percentage of tolerant organisms (4.8 percent) in the summer of 2005. Of the nine sites that had tolerant organisms present in both summers, seven had higher percentages in the summer of 2005 than 2004.

The percentage of dominant taxa is the percentage composition of the single most abundant taxa in a stream and is expected to increase in response to disturbance

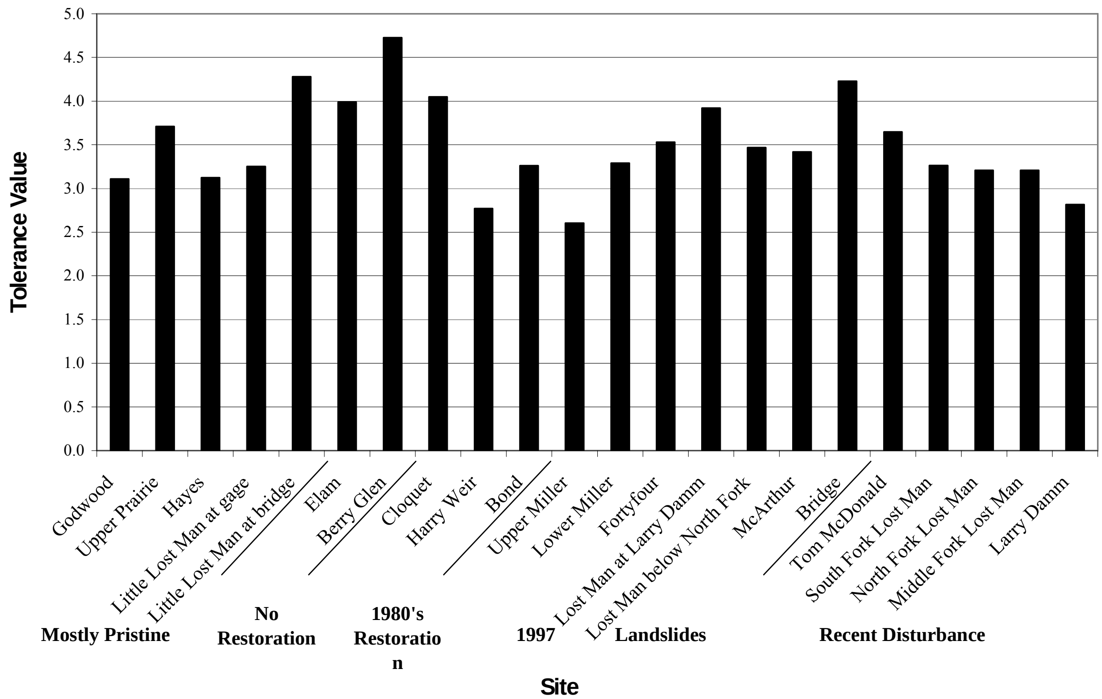


Figure 4-37. Spring macroinvertebrate tolerance values sampled from tributaries of Redwood Creek in 2005 with a 500µm benthic kick net.

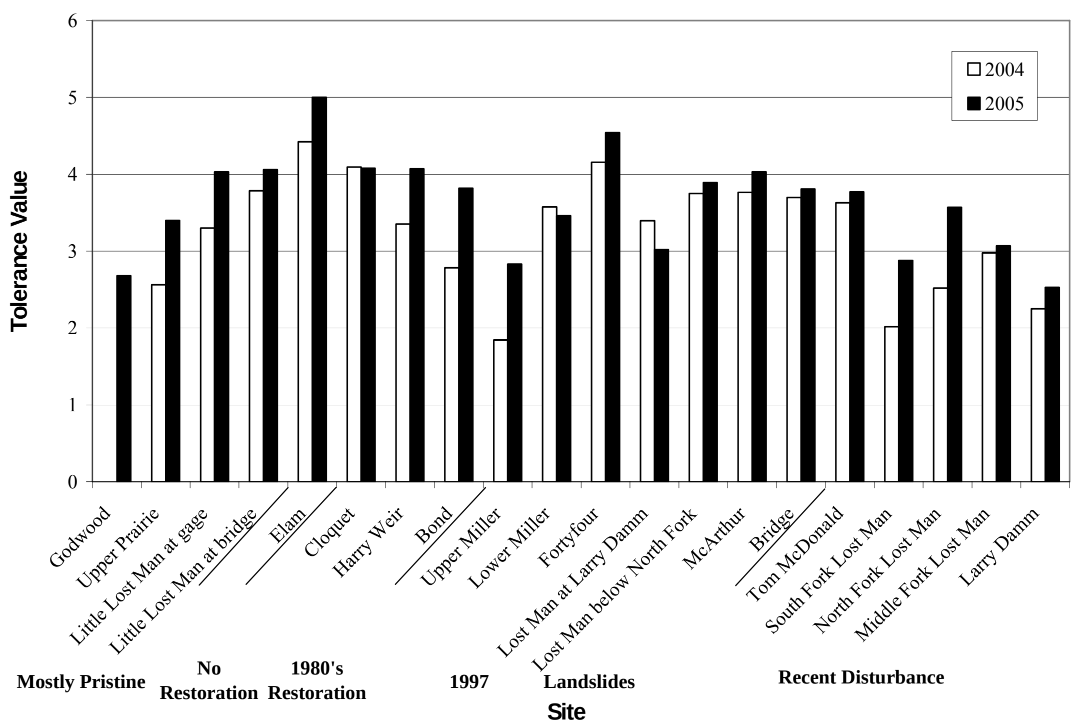


Figure 4-38. Summer macroinvertebrate tolerance values sampled from tributaries of Redwood Creek in 2004 or 2005 with a 500µm benthic kick net.

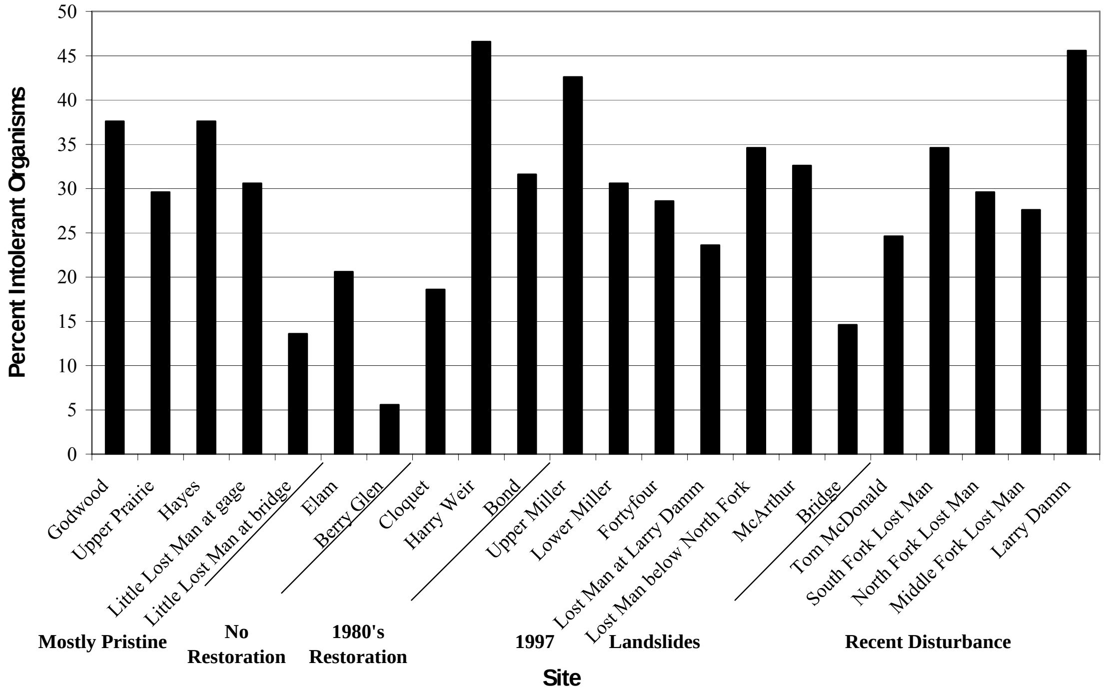


Figure 4-39. Spring percentage of intolerant macroinvertebrates sampled from tributaries of Redwood Creek in 2005 with a 500µm benthic kick net.

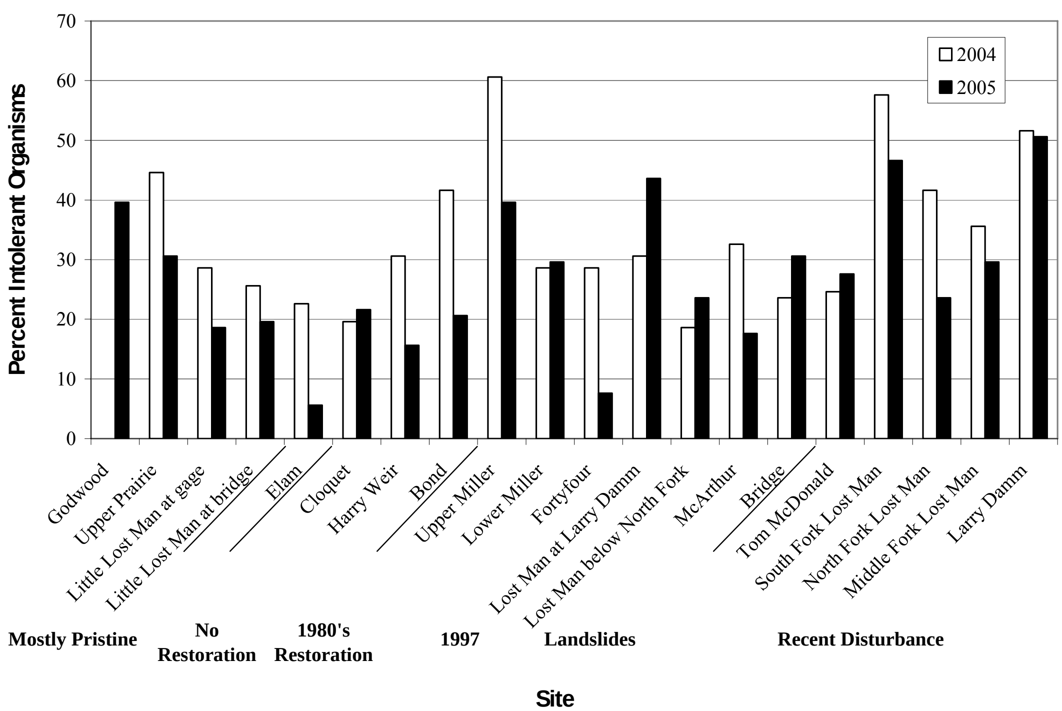


Figure 4-40. Summer percentage of intolerant macroinvertebrates sampled from tributaries of Redwood Creek in 2004 or 2005 with a 500µm benthic kick net.

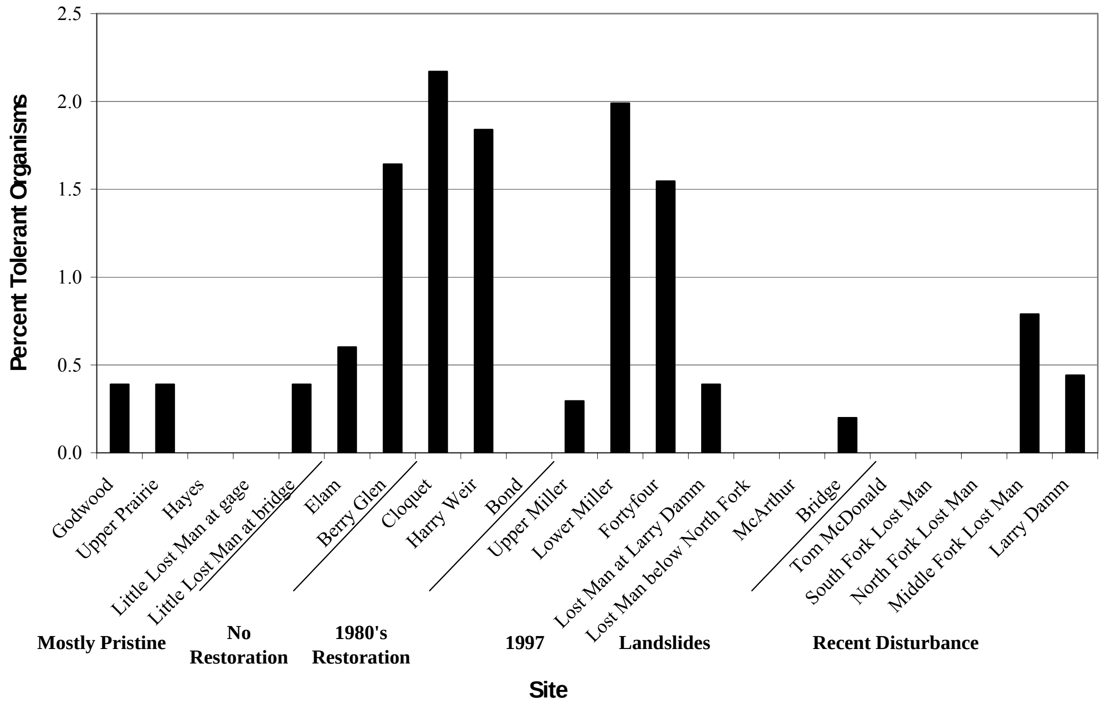


Figure 4-41. Spring percentage of tolerant macroinvertebrates sampled from tributaries of Redwood Creek in 2005 with a 500µm benthic kick net.

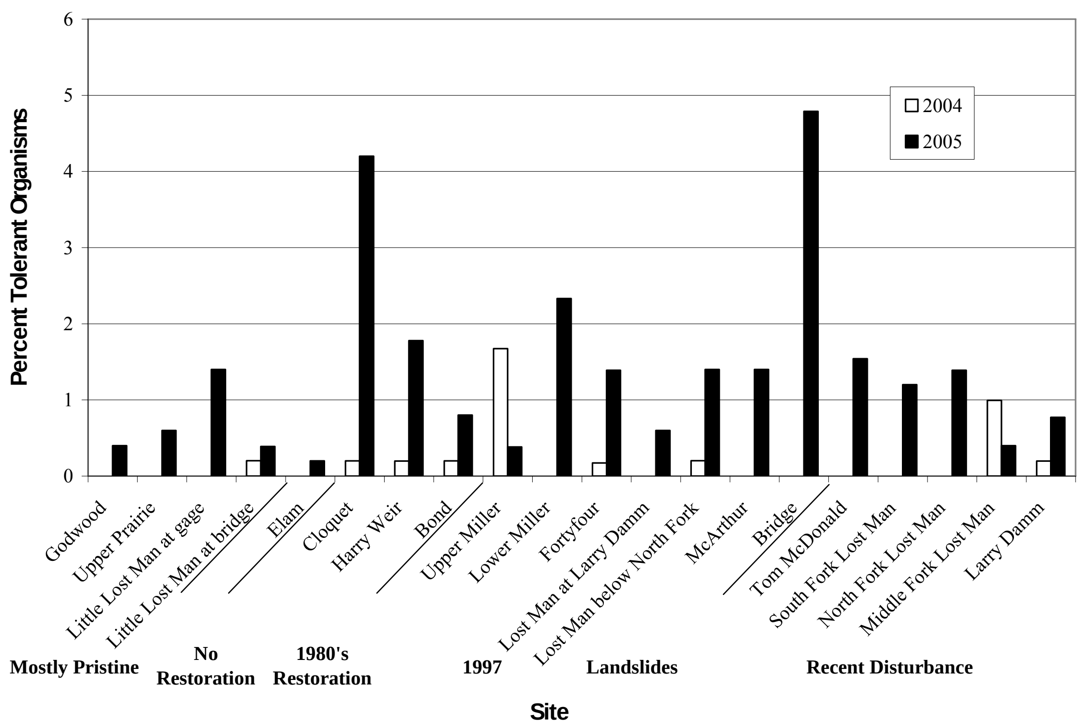


Figure 4-42. Summer percentage of tolerant macroinvertebrates sampled from tributaries of Redwood Creek in 2004 or 2005 with a 500µm benthic kick net.

(Harrington and Born, 2000). The spring percentage of dominant taxa was highest in Berry Glen Creek (Amphipoda) and lowest in Upper Miller Creek (Epeorus and Baetis) (Figure 4-43). All streams had less than 50 percent dominant taxa in the spring. The percentage of dominant taxa was lower in the summer of 2004 than 2005 in 15 of the 19 sites sampled in both years (Figure 4-44). With the exception of Bond and Fortyfour Creeks in 2005, all sites had less than 50 percent dominant taxa in both summers.

Benthic macroinvertebrates collected following the rapid bioassessment protocol were placed into FFG's derived from lists developed for the Pacific Northwest by Aquatic Biology Associates, Inc (CAMLnet, 2003). FFG's were as follows: predators, parasites, collector-gatherers, collector-filterers, macrophyte herbivores, piercer herbivores, scrapers, shredders, omnivores and xylophages. When presenting the proportional bioassessment metrics, only filtering-collectors, gathering collectors, scrapers, shredders and predators are shown because they make up the majority of the composition of the streams. The other FFG's are presented in Appendices 4-3 - 4-10.

The percentage of gathering-collectors generally made up the majority of aquatic invertebrates at all sites. During the spring of 2005, Berry Glen Creek had the highest percentage of gathering-collectors (73 percent) and Upper Miller Creek had the lowest (30 percent) (Figure 4-45). The percentage of gathering-collectors was higher in 2005 than 2004 in 15 of the 19 sites sampled during both summers (Figure 4-46). Elam and Fortyfour Creeks both had percentages of gathering-collectors reach over 80 percent in the summer of 2005. There did not appear to be a trend between level of disturbance and percentage of gathering-collectors.

The percentage of predators was highest (34 percent) in Larry Damm Creek and lowest in Middle Fork Lost Man Creek (9 percent) during the spring of 2005 (Figure 4-47). The percentage of predators was higher during the summer of 2004 than 2005 in 18 of the 19 sites sampled during both years (Figure 4-48). Lost Man Creek below North Fork had a higher percentage of predators during the summer of 2005 than 2004. The percentage of predators was five times greater in the summer of 2004 than 2005 in Fortyfour Creek. South Fork Lost Man Creek had the highest percentage of predators during both summers.

The spring and summer percentage of shredders stayed below 20 percent at all sites during all sampling periods. During the spring of 2005, the percentage of shredders was highest in Lower Miller Creek (18 percent) and lowest in Bridge Creek (less than 1 percent) (Figure 4-49). The percentage of shredders was higher in the summer of 2004 than in 2005 in 12 of the 19 sites sampled both years (Figure 4-50). Elam Creek had the highest percentage of summer shredders in 2004 and Lost Man Creek at Larry Damm had the lowest percentage in 2005.

The spring percentage of scrapers was highest in Harry Weir Creek (29 percent) and lowest in Berry Glen Creek (7 percent) during 2005 (Figure 4-51). The summer percentage of scrapers was higher in the summer of 2005 than 2004 in 13 of the 19 sites sampled during both years (Figure 4-52). The percentage of scrapers stayed below 5 percent during both summers in Elam and Fortyfour Creeks. Lost Man Creek at Larry Damm had the highest percentage (46) of scrapers during the summer of 2005.

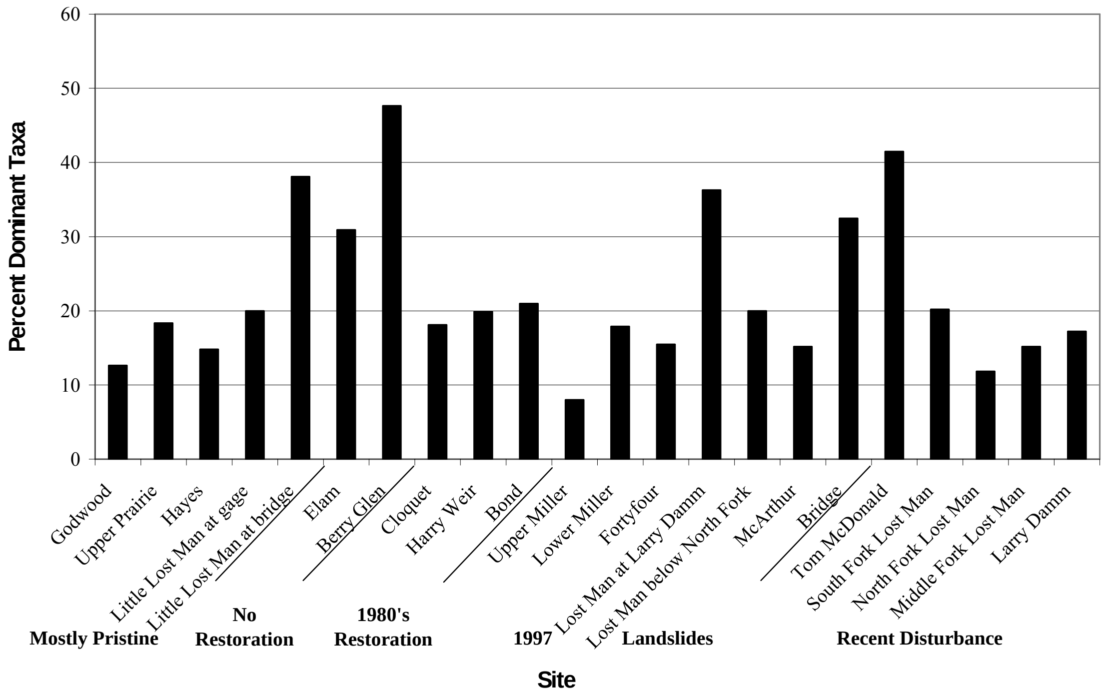


Figure 4-43 Spring percentage of dominant taxa sampled from tributaries of Redwood Creek in 2005 with a 500µm benthic kick net.

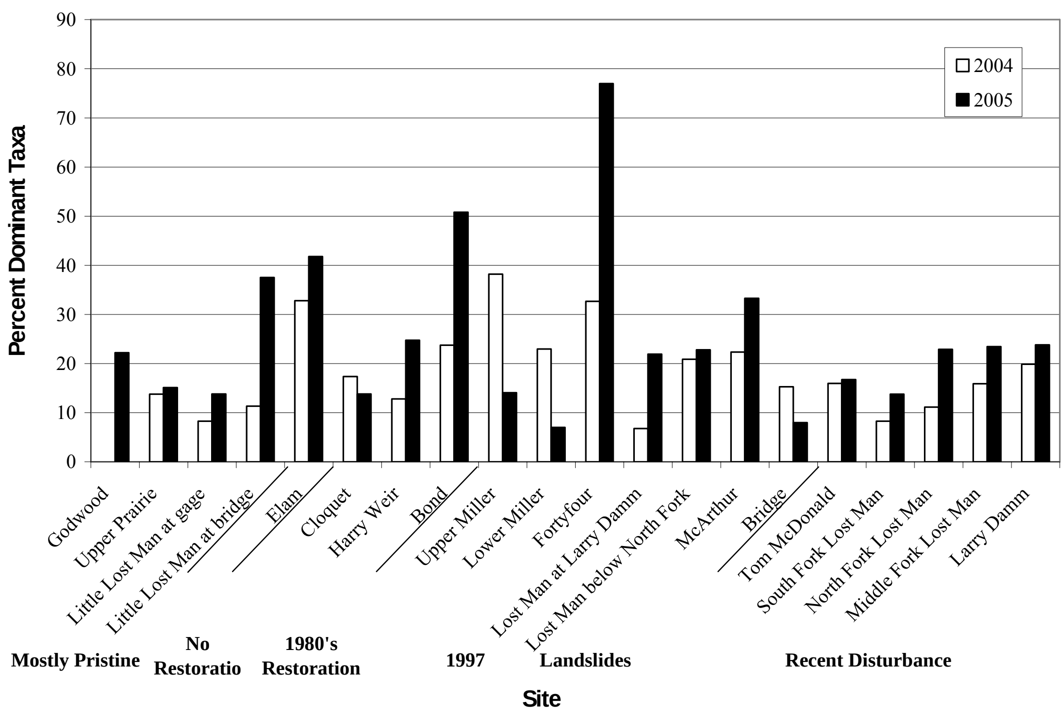


Figure 4-44. Summer percentage of dominant taxa sampled from tributaries of Redwood Creek in 2004 or 2005 with a 500µm benthic kick net.

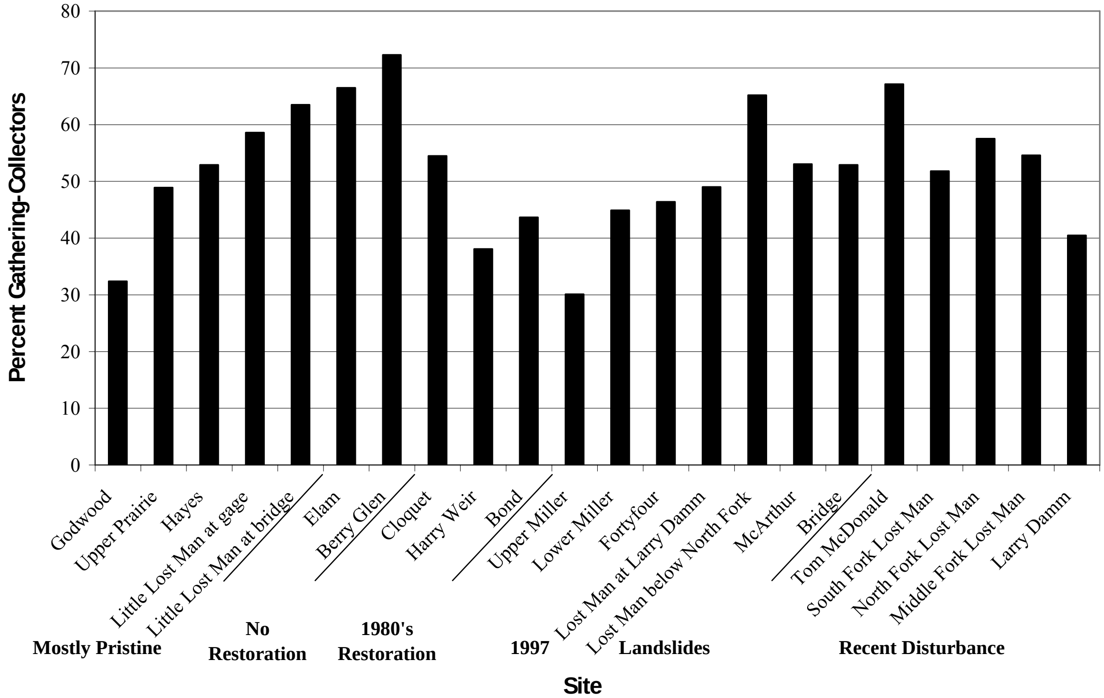


Figure 4-45. Spring percentage of gathering-collector macroinvertebrates sampled from tributaries of Redwood Creek in 2005 with a 500µm benthic kick net**.**

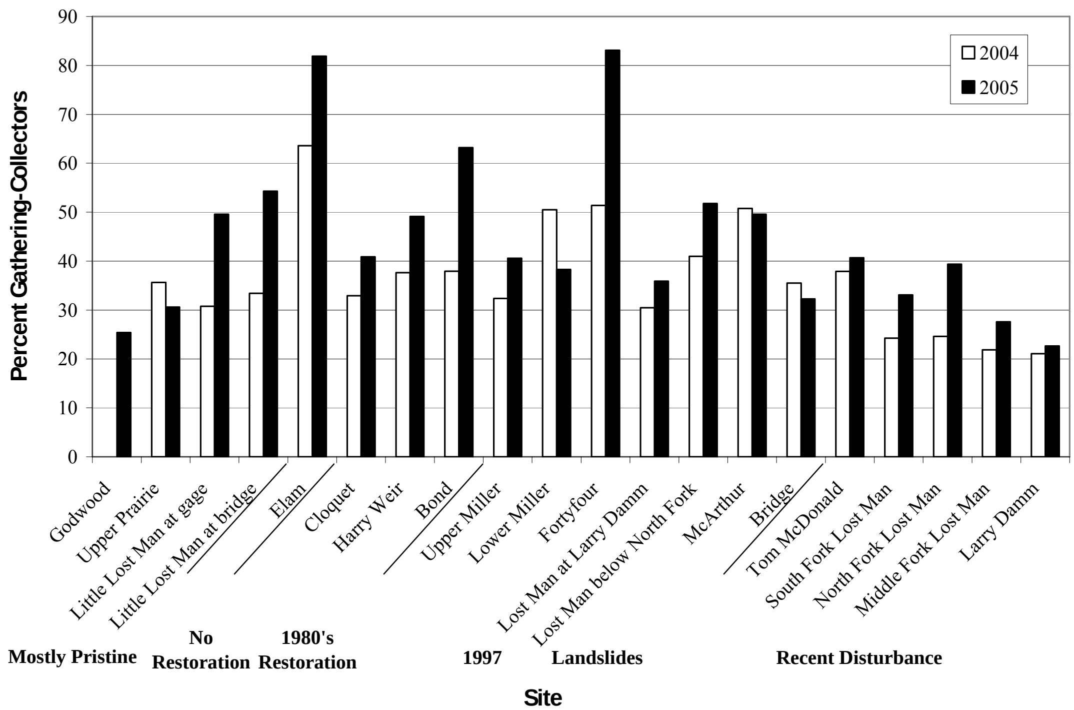


Figure 4-46. Summer percentage of gathering-collector macroinvertebrates sampled from tributaries of Redwood Creek in 2004 or 2005 with a 500µm benthic kick net**.**

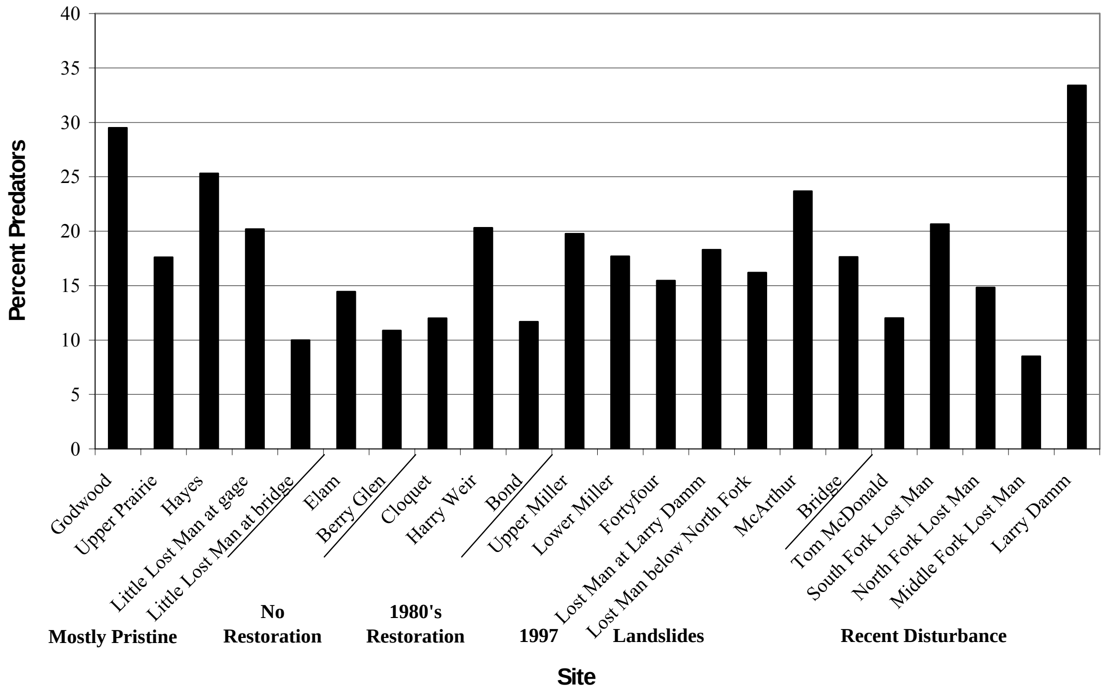


Figure 4-47. Spring percentage of predator macroinvertebrates sampled from tributaries of Redwood Creek in 2005 with a 500µm benthic kick net.

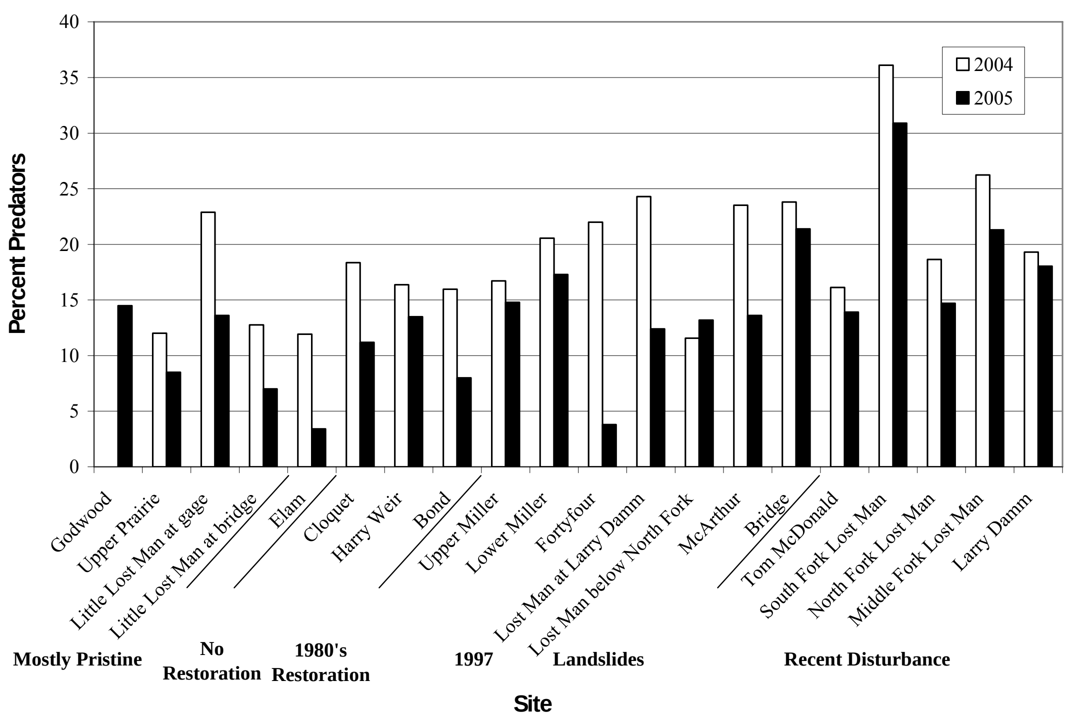


Figure 4-48. Summer percentage of predator macroinvertebrates sampled from tributaries of Redwood Creek in 2004 or 2005 with a 500µm benthic kick net.

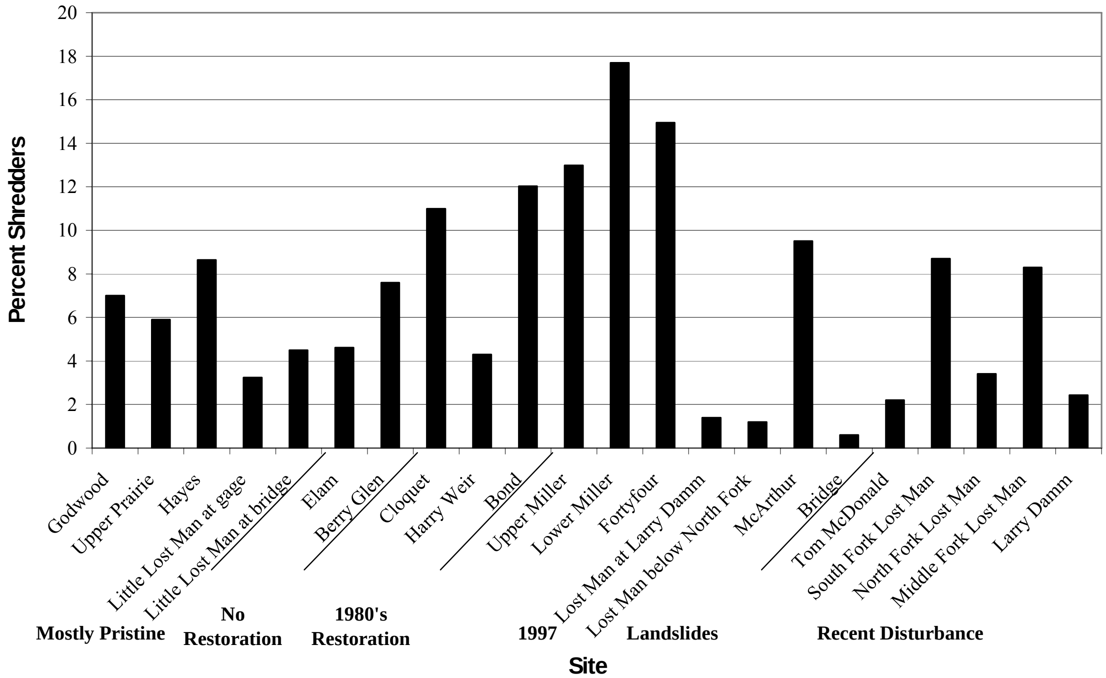


Figure 4-49. Spring percentage of shredder macronvertebrates sampled from tributaries of Redwood Creek in 2005 with a 500µm benthic kick net.

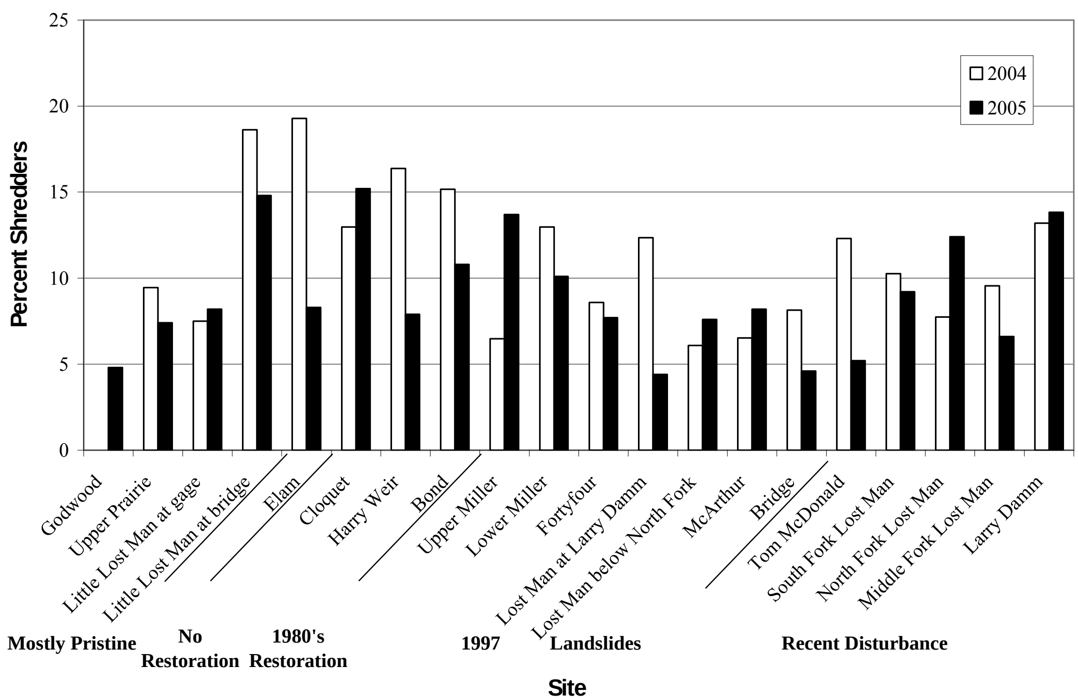


Figure 4-50. Summer percentage of shredder macroinvertebrates sampled from tributaries of Redwood Creek in 2004 or 2005 with a 500µm benthic kick net.

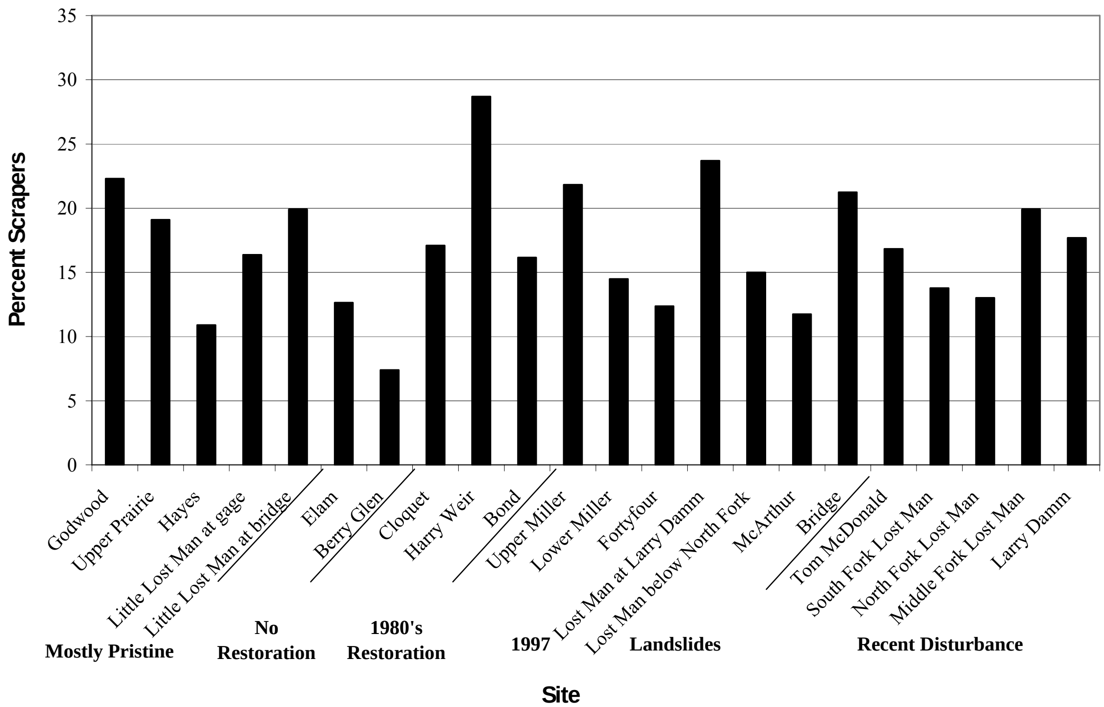


Figure 4-51. Spring percentage of scraper macroinvertebrates sampled from tributaries of Redwood Creek in 2005 with a 500µm benthic kick net.

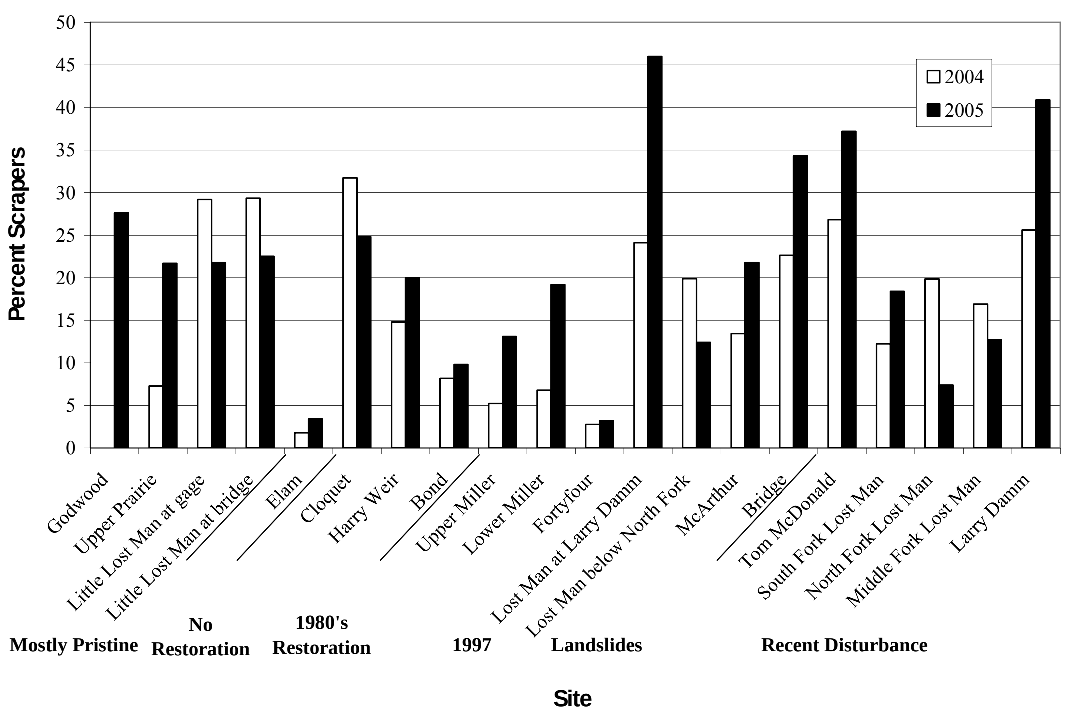


Figure 4-52. Summer percentage of scraper macroinvertebrates sampled from tributaries of Redwood Creek in 2004 or 2005 with a 500µm benthic kick net.

The spring percentage of filtering-collectors remained below 13 percent at all sites during 2005 (Figure 4-53). It was highest in Bond Creek (13 percent) and remained below 1 percent in Little Lost Man Creek at the gage, Lost Man Creek below North Fork and Berry Glen, McArthur and Tom McDonald Creeks. The summer percentage of filtering-collectors in 2004 was equal to or greater than that in 2005 in 12 of the 19 sites sampled both years (Figure 4-54). The summer percentage of filtering-collectors was highest in Middle Fork Lost Man Creek in 2005 and lowest in Upper Miller Creek in 2004.

The index of biotic integrity was calculated for sites sampled following the rapid bioassessment protocol based on eight metrics: number of EPT taxa, number of Coleopteran taxa, number of Dipteran taxa, percentage of intolerant organisms, percentage of non-Gastropod scrapers, percentage of predators, percentage of shredders and percentage of non-insect taxa. A single score is determined by adding the scores of the eight individual metrics. The final score is a single number describing the overall health of a stream. The final scoring range is: 0-20 = very poor, 21-40 = poor, 41-60 = fair, 61-80 = good and 81-100 = very good. The IBI used in this study is based on one that was created for southern California. The California Department of Fish and Game is currently developing an IBI specifically for northern coastal California, so the values of IBI in this study will need to be updated when the new IBI metrics are approved.

The spring IBI score was highest in Harry Weir and Lower Miller Creeks (69) and lowest in Berry Glen Creek (35) (Figure 4-55). The IBI score was higher in the summer of 2004 than 2005 in 13 of the 19 sites (Figure 4-56). The score was highest in Larry Damm Creek in the summer of 2005 and lowest in Fortyfour Creek in the summer of 2005. With the exception of Little Lost Man Creek at the bridge and Elam and Fortyfour Creeks, sites did not differ by more than ten points between the summers of 2004 and 2005.

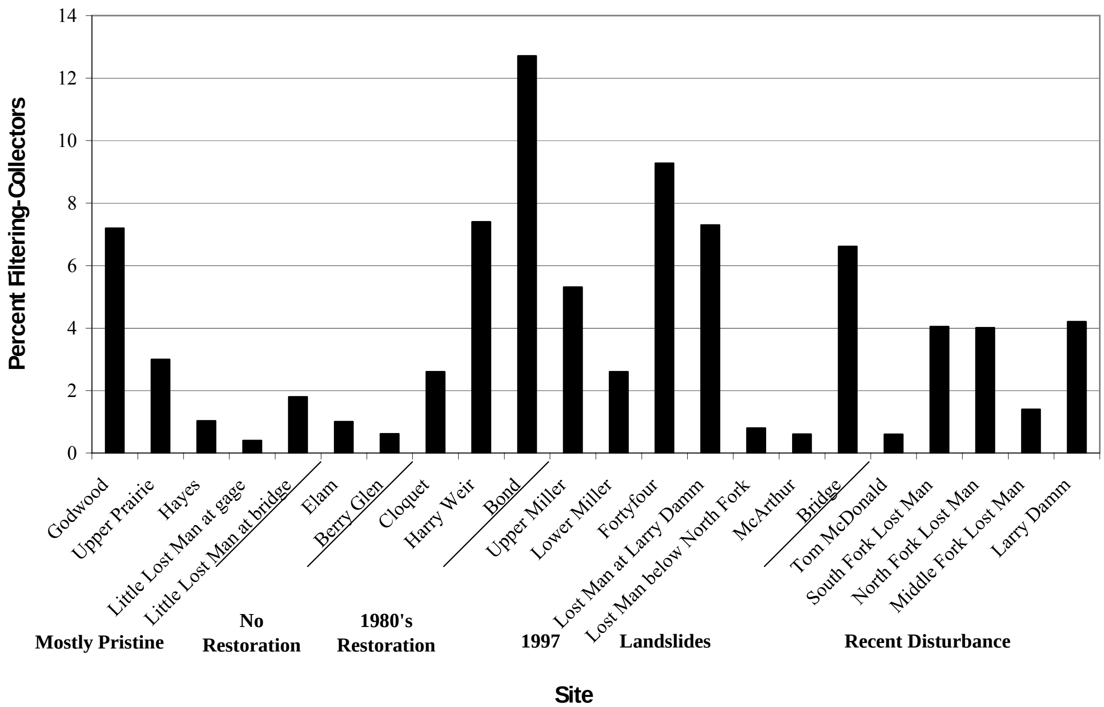


Figure 4-53. Spring percentage of filtering-collector macroinvertebrates sampled from tributaries of Redwood Creek in 2005 with a 500µm benthic kick net**.**

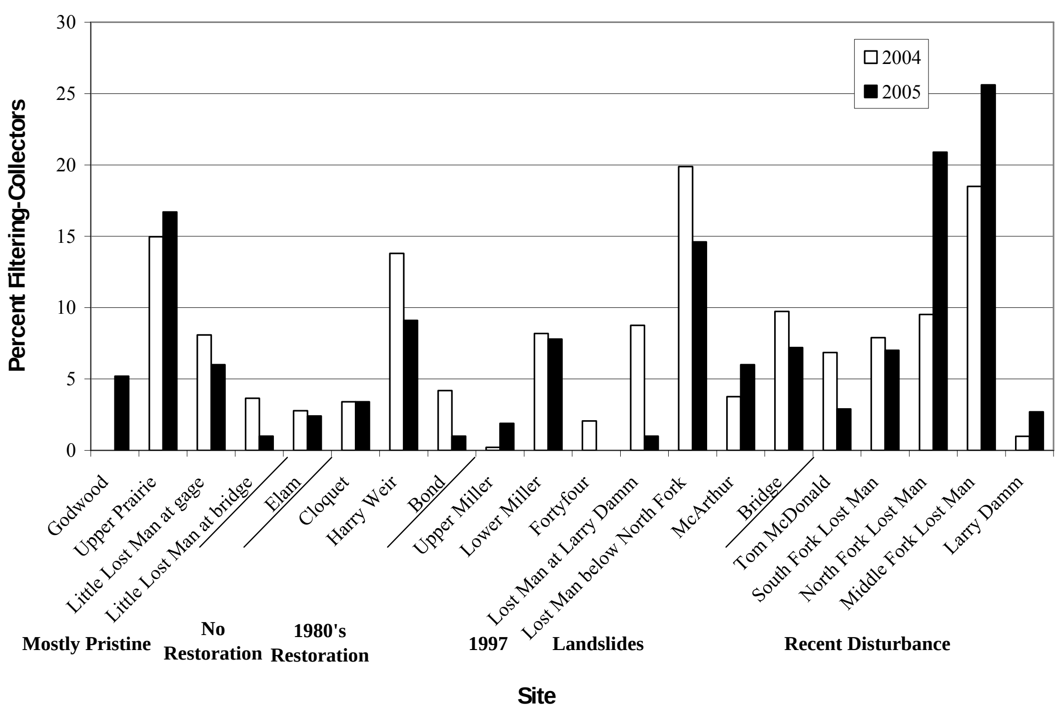


Figure 4-54. Summer percentage of filtering-collector macroinvertebrates sampled from tributaries of Redwood Creek in 2004 or 2005 with a 500µm benthic kick net**.**

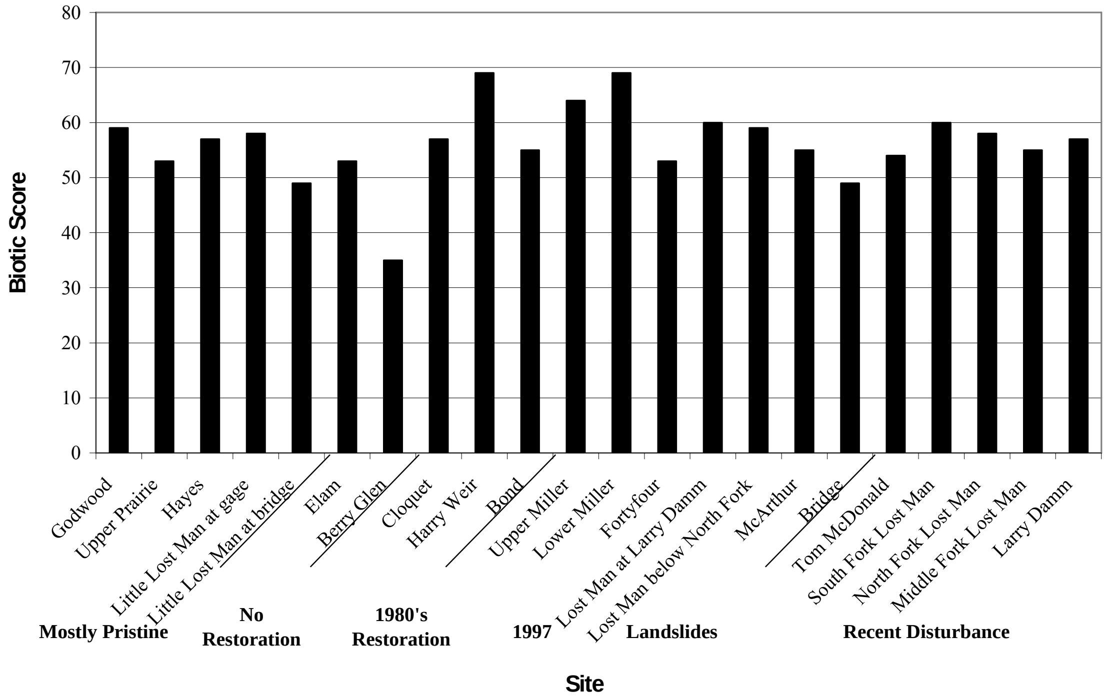


Figure 4-55. Spring macroinvertebrate index of biological integrity sampled from tributaries of Redwood Creek in 2005 with a 500µm benthic kick net.

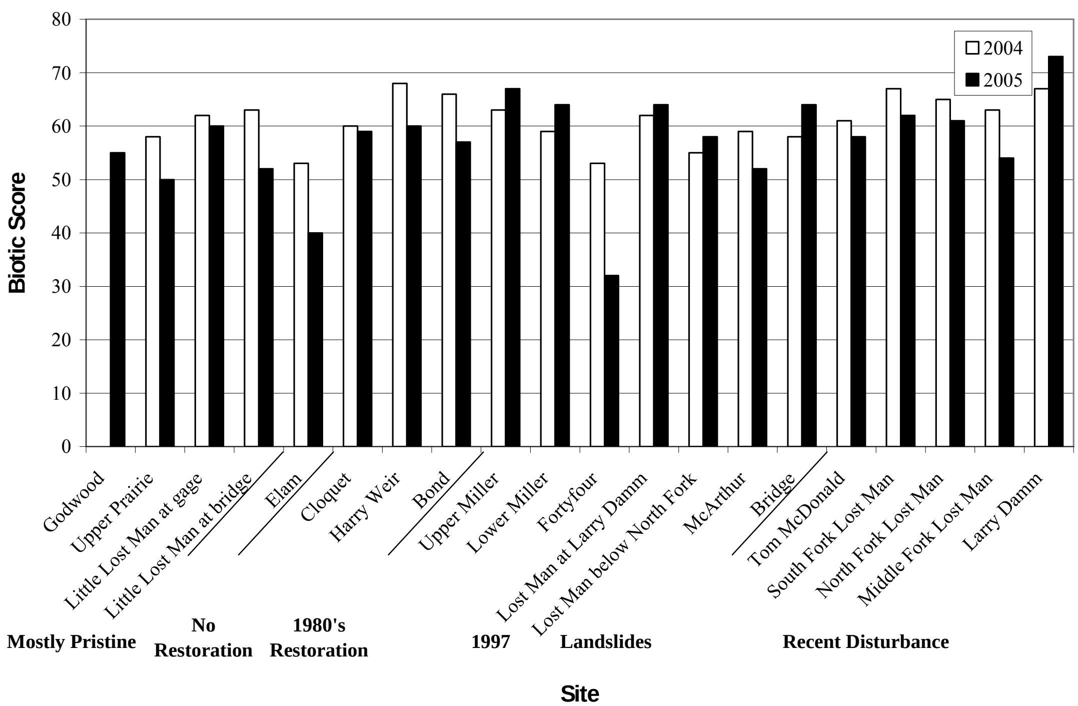


Figure 4-56. Summer macroinvertebrate index of biological integrity sampled from tributaries of Redwood Creek in 2004 or 2005 with a 500µm benthic kick net**.**

Discussion and Conclusions

Surber samples

A flood with a recurrence interval of about 25 years occurred in March, 1975, and so some of the interannual variation in densities may be due to flood effects. Discharge in the spring of 1975 was three times higher than in the spring of 1974, yet invertebrate densities were higher in the spring of 1975 than 1974 in six of the eight sites sampled in both years. Flooding events often lead to scouring of the streambed and a decrease in macroinvertebrate abundance, with pre-flood levels recovering within two to three months following the disturbance (Shannon and others, 2001 and Angradi, 1997). Spring sampling occurred in mid-May in 1974 and at the very end of May and the beginning of June in 1975. This may account for some of the differences in macroinvertebrate abundance, although this is not consistent with a two to three month recovery period. Angradi (1997) found that shifts in community structure following floods were small compared to seasonal variation. It is possible that seasonal variation between the two years of study could also play a greater role in abundance than did a high flow event.

We originally hypothesized that streams which were highly disturbed in the 1970's would have an increase in diversity in 2004-2005, but this was not the case. Some insects respond positively to certain types of disturbance. For example, when a stream receives more light due to removal of the canopy, periphyton may increase and numbers of scrapers will likewise increase. The percentage of scrapers may also have been more abundant in the summer than in the spring because benthic periphyton tends to be more abundant in the summer months.

Although there were not clear trends in all the bioassessment metrics, some generalizations can be made. Filtering-collectors filter fine particulate matter and are expected to increase in response to disturbance (Harrington and Born, 2000). Percentages of filtering collectors were higher in the spring of 1974 than in 2004 at most sites. Gathering collectors are the macrobenthos that collect or gather fine particulate matter and are expected to increase in response to disturbance (Harrington and Born, 2000). Late summer percentages of gathering collectors tended to be higher in 1975 than in 2004. Percentages of insects exhibiting 2+ year life cycles (indicative of more stable channel conditions) increased from 1974 to 2004.

Benthic Kick Samples

It should be noted that invertebrates from kick samples were identified by different biologists and so a small amount of taxonomic difference in identification may exist between samples. This mainly affects richness metrics because they take into account distinct taxa. Samples collected in 2005 were often taken to a lower taxonomic level, resulting in a higher number of distinct taxa, than those collected in 2004. These differences are reflected in richness metrics and the index of biotic integrity score, which takes into account three richness metrics when determining the final biotic score.

Karr and Chu (1999) point out that the taxa richnesses of Ephemeroptera, Plecoptera and Trichoptera reflect different types of degradation. Plecoptera tend to decline at less intense levels of human influence than Ephemeroptera or Trichoptera, therefore combining these three taxa into a single "EPT" metric may hide differences that could help diagnose both the types and sources of degradation at a site. Fore and others (1996) found that total taxa richness and richnesses of Ephemeroptera, Plecoptera and Trichoptera easily separated the degree to which sites had been disturbed by logging and road construction activities and declined as disturbance increased. They also found Plecoptera to be more sensitive to human influence because they disappeared from sites at lower levels of disturbance than did either Ephemeroptera or Trichoptera.

During the spring of 2005, both Ephemeroptera and Plecoptera richness were highest in South Fork Lost Man Creek, which has been recently disturbed. Trichoptera richness was highest in Upper Prairie Creek during the spring, which is considered a mostly pristine site. During the summers of 2004 and 2005, Ephemeroptera taxa richness was highest in Bridge Creek in 2005, a site with landslides in 1997, Plecoptera richness was highest in Middle Fork Lost Man Creek in 2005 and Trichoptera richness was highest in South Fork Lost Man Creek in 2004, both recently disturbed sites. Unlike the studies mentioned above, Plecoptera taxa richness was highest in recently disturbed sites in both spring and summer. There does not appear to be a relationship between the level of watershed disturbance and Ephemeroptera, Plecoptera or Trichoptera taxa richness in 2004 or 2005.

Some generalizations can be made based on both Surber and kick sample results. Large floods in 1975 and 1997 and erosional disturbances (landslides, road failures, and adjustment of restored stream crossings) probably influenced the abundance and diversity of invertebrates, but there was not a clear trend in bioassessment metrics and the degree of disturbance. Based on the ratio of scrapers to shredders and total collectors, most of the sampled streams were heterotrophic (they are dependent upon allochthonous organic inputs more than autochthonous primary production). The ratio of filtering collectors (FG) to gathering collectors (GC) tended to be higher in streams that were mostly pristine or were not recently disturbed. In general, long-lived species (those with lifecycles longer than a year) are more abundant in recent years than in the 1970's.

A limitation of this study was that we needed to repeat the sampling conducted by the USGS in the 1970's during the same season as the earlier sampling. The timing of the original sampling (late spring and early fall) was not optimum in terms of macroinvertebrate life cycles. In the future, sampling in July to identify summer generations and in March to sample winter generations may provide more unambiguous results. Another limitation is that the sampling in previous studies and in the current study only tested riffle habitat. In this region, as a stream recovers, it is the pool habitat that changes to a greater degree (Madej, 1999). Sampling targeted to slow-water habitats may yield different results than those from fast-water riffles.

Similar aquatic macroinvertebrate studies are ongoing by other groups in the area. As compared to studies by the Yurok Tribe in the spring of 2003 and 2004, the percentage of dominant taxa, sensitive EPT index and tolerance values were similar to our current results (Yurok Tribe Environmental Program 2004 and 2005). The taxa richness and EPT taxa richness found in the Yurok sites were slightly lower in 2003 than in the current study, but were similar in 2004. Studies done in streams on timberlands owned by The Pacific Lumber Company showed that tolerance values and EPT taxa richness were slightly lower than those found in the current study, the percentage of scrapers was slightly higher and the percentage of dominant taxa was similar (R2 Resource Consultants, 1996). Similar studies are also currently being conducted in the

Freshwater Creek watershed by Humboldt State University's Institute for River Ecosytems, and a regional comparison of bioassessment metrics should soon be possible.

In some studies of disturbed watersheds, disturbance, such as urbanization, is considered a single factor and stream reaches with different levels of urbanization are sampled. In this approach biotic assemblages at the sampling sites are analyzed in terms of that single factor to infer effects of that disturbance (Brown and others, 2005). Because in the Redwood Creek basin, 'disturbance' cannot be stated in terms of a single factor, indices of biotic integrity do not follow a simple pattern of lowered integrity with increased disturbance. The constraint of this retrospective study is that the original study sites were not randomly selected and treatments were not conducted within an experimental framework. To further elucidate the effects of road removal on the aquatic biotic assemblages, a more stringent monitoring design would be necessary.

Chapter 5 Amphibians

Introduction

Amphibians can serve as an appropriate vertebrate indicator to assess the health of local aquatic conditions because they display limited dispersal (Daugherty and Sheldon, 1982a; Welsh and Lind, 1992), are relatively long lived (10+ yrs), (Daugherty and Sheldon, 1982b) and have an aquatic and terrestrial life history. Corn and Bury (1989), and Welsh and Ollivier (1998) have found stream amphibians to be negatively affected by landscape disturbance activities. In particular, studies have reported that tailed frogs are sensitive to fine sediments (Corn and Bury, 1989; Ashton and others, 2006) and are a reliable environmental indicator species (Welsh and Ollivier, 1998).

Prior to the formation and expansion of Redwood National Park (RNP) in 1968 and 1978, respectively, nearly 75 percent of the current parkland was logged by private timber companies (Best, 1995). Redwood National Park began a long-term watershed restoration program in the late 1970's to decrease sediment input from roads into streams and improve stream habitat for aquatic biota (Madej, 2001a). To date, nearly 300 km of roads have been removed in RNP.

We conducted a retrospective study to investigate the effects of recent (e.g. road removal) and legacy (e.g. historic timber harvests and road construction) disturbance events on the abundance of larval amphibians in third and fourth order tributaries to Redwood Creek, California. The objectives were to determine whether the level of road removal work conducted in select sub-basins of Redwood National and State Parks (RNSP) had an effect on stream amphibian abundance.

Methods

We sampled 14 study reaches located in 13 third- and fourth-order (based on 1:24000 topographic maps) tributaries to Redwood Creek, RNSP. A 100-m reach was delineated at each study site about 50 -100 m upstream from the confluence with Redwood Creek or Prairie Creek and located more than 200 m downstream from all road removal work. The study sites were located in sub-basins classified into one of three levels of disturbance based on RNSP road removal work: no road removal and mostly unlogged (NR) — Hayes, Little Lost Man at bridge, Little Lost Man at gage, and Upper Prairie Creeks; moderate level of road removal (MR)—Bond, Fortyfour, Harry Weir, McArthur and Lower Miller Creeks; and high level of road removal (HR)—Bridge, Larry Damm, Lost Man, and Tom McDonald Creeks. Moderate road removal was defined as sub-basins with less than 5000 m^3 of excavated road fill/km^2 of basin area, with road removal conducted primarily in the 1980's and early 1990's. High road removal basins had 5300 to 49000 5000 m^3 of excavated road fill/km^2 of basin area, with road removal conducted primarily in the late 1990's and 2000's.

Sites were sampled in a three-week period in late May to mid June 2004 and late April through early June 2005. We used the single habitat approach sampling methodology (i.e., riffle/run) (Bury and Corn, 1991). A two-person crew sampled six randomly selected 2-m belts in fast water habitats within each 100-m reach for five minutes/belt. If a randomly chosen belt was too deep (> 50 cm) or unsafe to sample, the next closest belt was sampled in its place.

We identified all amphibians to species in the field and measured mass to the nearest 0.1 g. We measured several habitat variables at each belt: 1) stream wetted width, 2) average depth, and 3) percentage of substrate composition (fines, small gravel, cobble, boulder, bedrock). Additional habitat variables measured at the reach level were particle size distributions through pebble counts, S* (a measure of fine sediment load per stream reach) and stream gradient. We used a geographical information system (ArcMap) to identify the sub-basin variables: 1) stream order, 2) percentage of old growth, 3) sub-basin drainage area, 4) road density, 5) year of last timber harvest, 6) year of last road removal work, 7) length of roads removed (km/ km^2), and 8) cubic meters excavated at stream crossings/ km^2. We used ANOVA (Sigmastat Software v 3.1 help) to compare differences for larval tailed frog biomass and densities across disturbance categories within years. Results were judged to be statistically significant at $p \leq 0.05$.

Results

Three amphibian species were captured during the study: tailed frogs (*Ascaphus truei*), Pacific giant salamanders (*Dicamptodon tenebrosus*) and foothills yellow- legged frogs (*Rana boylii*). The NR study sites contained the mean highest abundance of tailed frogs for 2004 and 2005 (Table 5-1). Little Lost Man at the gage, Upper Prairie, Bond, McArthur, and Tom McDonald study sites had similar levels of tailed frogs captured across both study years. For the Little Lost Man at bridge, Forty-four, and Lost Man sites, there was a two to threefold increase in tailed frog captures for 2005 compared to 2004. For the Harry Weir, Miller, and Larry Damm sites there was a two to threefold decrease in tailed frog captures for 2005 compared to 2004. In general, fewer Pacific giant salamanders were captured at all study sites during 2005 compared to 2004. Foothills yellow-legged frogs were only found at three sites during the course of the study.

2004

A total of 231 tailed frogs, 79 Pacific giant salamanders, and two foothills yellow-legged frogs were captured during 2004 (Table 5-1). Mean tailed frog abundance for 2004 was 4.00 (± 0.82), 3.53 (± 0.62), and 2.30 (± 0.50) per five-minute belt survey for NR, MR, and HR sites, respectively. There was not a significant difference for larval tailed frog biomass between MR (1.90 ± 0.36 gms/m^2) and HR (1.50 ± 0.40 gms/m^2) study sites (p= 0.2344). The biomass of tailed frogs in NR (3.66 ± 0.80 gms/m^2) was significantly higher than at HR, but not as high compared to MR study sites (p=0.0074 and 0.0768, respectively). Tailed frog biomass was similar between MR and HR sites but approximately two times greater in NR compared to MR and HR sites (Figure 5-1). We found a trend towards significance for larval frog densities between MR (2.86 ± 0.49 frogs/m^2) and HR (1.53 ± 0.34) study sites (p = 0.0392) but not between NR (2.65 ± 0.53) and MR study sites (p=0.8685). There was a significant difference for larval frog density levels between NR versus HR study sites (p = 0.0485). In general, larval tailed frog densities were higher at NR and MR compared to HR sites (Figure 5-2).

Table 5-1. Abundance of tailed frogs, Pacific giant salamanders, and foothills yellow-legged frogs in 2004 and 2005 at select study sites located in the Redwood Creek watershed, Redwood National and State Parks, California. NS = not sampled, NR = no road removal and mostly pristine, MR = moderate road removal, HR = high road removal.

Study Site	**Road removal category**	**Tailed frog**		**Pacific giant salamander**		**Foothills yellow - legged frog**	
		2004	***2005***	***2004***	***2005***	***2004***	***2005***
Godwood	NR	NS	40	NS	9	NS	0
Hayes	NR	NS	47	NS	8	NS	2
Little Lost Man at bridge	NR	23	61	9	5	0	0
Little Lost Man at gage	NR	38	48	14	4	0	0
Upper Prairie	NR	11	16	10	10	0	0
Bond	MR	33	32	3	2	1	0
Forty-four	MR	5	7	2	1	0	0
Harry Weir	MR	15	6	5	0	0	0
McArthur	MR	20	18	6	0	1	0
Lower Miller	MR	33	15	7	0	0	0
Larry Damm	HR	10	6	14	8	0	0
Lost Man	HR	15	68	5	3	0	0
Tom McDonald	HR	28	34	1	0	0	0
Bridge	HR	0	NS	3	NS	0	NS

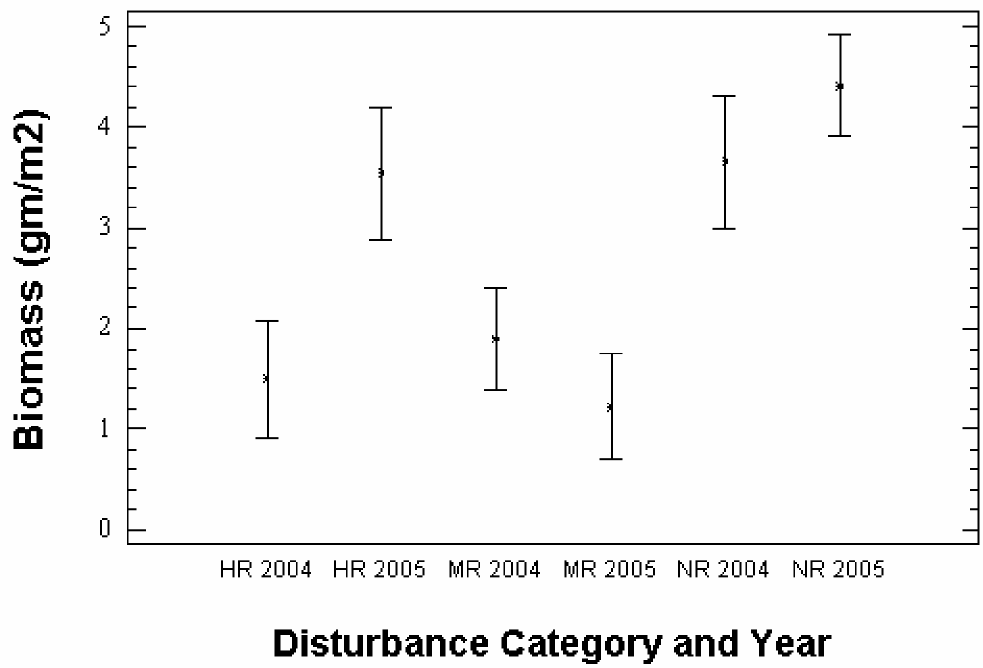


Figure 5-1. Mean tailed frog biomass in 2004 and 2005 for study reaches located in no road removal and mostly unlogged (NR), moderate road removal (MR), and high road removal (HR) sub-basins of Redwood Creek (error bars ± 1 SE).

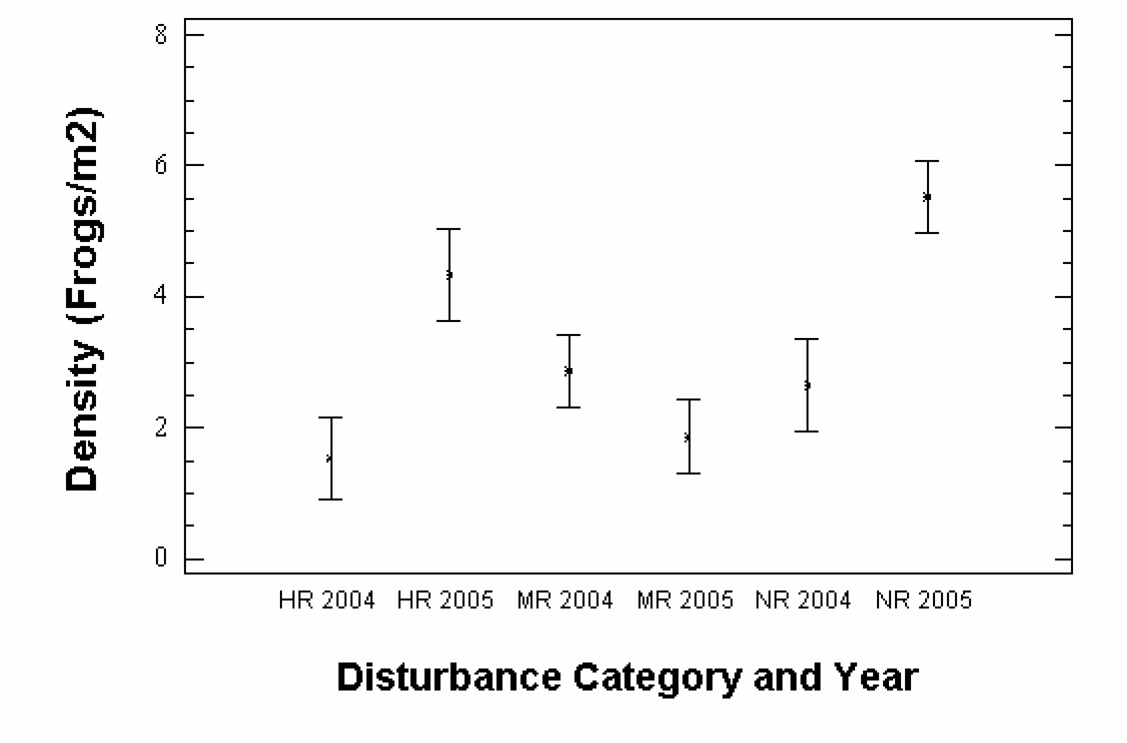


Figure 5-2. Mean tailed frog densities in 2004 and 2005 for study reaches located in no road removal and mostly unlogged (NR), moderate road removal (MR) and high road removal (HR) sub-basins of Redwood Creek (error bars ± 1 SE).

2005

A total of 398 tailed frogs, 50 Pacific giant salamanders, and two foothills yellow-legged frogs were captured during 2005. Mean tailed frog abundance for 2005 was 7.07 (± 0.79), 2.79 (± 0.58), and 6.00 (± 1.32) per five-minute belt survey for NR, MR, and HR sites, respectively. There was not a significant difference for larval tailed frog biomass between MR (1.22 ± 0.25 gm/m²) and HR (3.54 ± 0.90 gm/m²) study sites (p = 0.1011). The biomass of tailed frogs in NR (4.41 ± 0.70 gm/m²) was significantly higher than at MR, but not at HR study sites (p=0.0004 and 0.1148, respectively). Tailed frog biomass was approximately four times greater in NR compared to MR sites and 25 percent greater in NR versus HR sites (Figure 5-1). There was not a significant difference for larval tailed frog densities between MR (1.86 ± 0.37 frogs/ m²) and HR (4.34 ± 1.04) study sites (p=0.1252). The density of larval frogs in NR (5.52 ± 0.74 frogs/ m²) was significantly higher than at MR, but not as high compared to HR study sites (p=0.0004 and 0.0900, respectively).

Discussion and Conclusions

Amphibians were not sampled in the original 1974-75 USGS study. Amphibians were sampled in the Redwood Creek basin by USGS researchers in 1994 (Fellers, USGS, personal communication), but those surveys were not concentrating on the small tributary streams used in the present study. Consequently, the present study compared streams in sub-basins with and without road removal work, but could not detect changes through time. Nevertheless, this work does provide baseline data needed for future studies.

The biomass of tailed frogs in unlogged basins was significantly higher than in basins with moderate or high levels of road removal. Tailed frog densities were also higher in unlogged basins than in basins with moderate or high levels of road removal. These conclusions are consistent with those of Ashton and others (2006). They found that in redwood-dominated systems, recovery of headwaters amphibian assemblages may be suppressed for many decades after timber harvest even after recovery of the forest canopy. In 2004 and 2005 there was not a significant difference for larval tailed frog biomass between basins with moderate and high levels of road removal.

Chapter 6 Fish

Introduction

Fish production in freshwater streams is largely dependent on the quality of habitat available and this is most pronounced for salmonid species, which require relatively pristine streams in which to carry out a portion of their life cycles. Management of riparian areas greatly influences the quality of aquatic systems. Increased water temperatures, increased sedimentation, decreased dissolved oxygen concentrations and changes in channel structure are all results of land management activities adjacent to streams that can negatively impact salmonid populations. Timber harvesting and road construction are activities that generate sediment that reaches streams through surface erosion and mass movements, which can be detrimental to the survival of salmonids (Meehan, 1991).

Hicks and others (1991) describe four potential changes in the quality of salmonid habitat brought on by timber harvest from hillslopes and forest road construction: altered streamflow regimes, accelerated surface erosion and mass wasting, increased nutrient runoff and increased number of road crossings. Increased fine sediment in streams decreases the amount of oxygen available to fish and reduces spawning success and food abundance and results in a loss of winter hiding space while increases in coarse sediment can alter channel morphology and lead to increased or decreased rearing capacity. Increases in large woody debris alter channel structure and can block salmonid migrations and both reduce and increase cover. Increased nutrient runoff (mostly nitrates) leads to elevated nutrient levels in streams which increase autotrophic production and provides an increase in food available to salmonids, although this increase may be temporary. Road crossings, mostly bridges and culverts, provide barriers to upstream movement and can lead to increases in fine sediment input from road surfaces.

Coho salmon (*Oncorhynchus kisutch*) populations in northern California have been listed as endangered under the Endangered Species Act and are therefore closely monitored in northern California. Redwood Creek and its tributaries have historically supported both anadromous and resident salmonids. Based on surveys in 1974 and 1974, Averett and Iwatsubo (1981) reported that steelhead trout (*Oncorhynchus mykiss)* accounted for 69.6 percent of fish fauna captured (n = 1,066) during their study of Redwood Creek and its selected tributaries. They concluded, based upon ordinary least squares regression techniques of length and weight data, that the steelhead trout from the Redwood Creek drainage basin were substantially "slimmer" than other steelhead trout populations representative of small California coastal streams.

There have been several studies conducted within the boundaries of Redwood National Park in which length and weight data for juvenile steelhead trout were collected (unpublished data, from Redwood National and State Parks). These data were used in an attempt to assess any potential changes in juvenile steelhead trout condition in response to 30 years of watershed rehabilitation. The objectives of this study were to: 1) assemble and compare data collected from previous studies on the types and characteristics of fish occurring in tributaries of Redwood National Park, 2) use snorkel surveys to determine the presence and type of salmonids in tributaries of Redwood Creek, and 3) calculate and compare the condition of steelhead populations collected from tributaries in the Park over the past few decades.

Methods

Snorkel Survey

Snorkel dive surveys were conducted at 20 sites in July of 2004 and 22 sites in July of 2005 in order to determine the density and presence or absence of juvenile salmonids. Pool and run habitats greater than 0.1 m in depth within each 100 m study reach were dove by one diver on all occasions with the exception of units greater than 5m in width. These units were snorkelled by two divers. Surveys began at the downstream end of each reach and progressed upstream to the end. The total number and species of fish observed in each habitat were recorded using a hand counter. The length, width and depth of each unit snorkeled were recorded. Habitat unit visibility was given a subjective rating of poor, fair, or good by the diver.

Fish Condition

Length (mm) and weight (g) data for steelhead trout *(Oncorhynchus mykiss)* were used to compare recent fish condition to that documented 30 years ago for selected study reaches. Both sets of data were collected from surveys conducted in tributaries to Redwood Creek that included Bridge Creek, Little Lost Man Creek, Harry Weir Creek and Tom McDonald Creek and in the mainstem Redwood Creek near Redwood Valley. Fish collected during 1974 and 1975 surveys were collected via back-pack electrofisher and seine (Iwatsubo and others, 1976). Fish collected during 1994, 1995 and 2005 were collected solely by electrofishing techniques (unpublished data, Redwood National and State Parks) and fish collected during 2000 in Redwood Creek were collected by a rotary screw trap (unpublished data, California Department of Fish and Game). The relationship between the mean length and weight of two fish populations can be used to compare fish health across both spatial and temporal scales. Fish condition indices are often calculated using Fulton's condition factor, K and ordinary least squares regression techniques (Cone, 1989).

In 1975, the USGS assessed fish condition using length-weight relations for steelhead trout with the following equation (Iwatsubo and Averett, 1981),

$$W=aL^b$$

where W=weight in grams
L=length in mm
a=constant, and
b=slope of the regression line (Lagler, 1969):

The values of a and b were determined empirically from the actual fork length and the weights of the salmonid fish captured. The slope of the regression line, b, was used to indicate the extent of growth occurring in the salmonid fish captured; that is, the steeper the slope of the regression line, the more weight the fish is gaining per unit growth in length. Generally, when the slope of the regression is greater than three, the fish are stout, and when the slope of the regression is less than three, the fish are slim (Iwatsubo and Averett, 1981). In order to compare data from the 1970's surveys with those conducted in selected creeks by Redwood National Park staff from 1994 to 2005, we used the same regression technique. Differences in the slopes and intercepts of the regression curves for different time periods were tested using a multiple slopes model of the linear regressions.

Results

Snorkel surveys found that coho salmon were not present in either 2004 or 2005 in Berry Glen, Bond, Harry Weir, Hayes, Middle Fork Lost Man, South Fork Lost Man, Upper Miller and Lower Miller Creeks (Table 6-1). Coho were not present in 2004 in Cloquet Creek and 2005 in Fortyfour Creek. Hayes Creek and Upper Miller Creek had no fish present in either 2004 or 2005. In 1974 and 1975 Tom McDonald, Bridge, Lower Miller, Harry Weir and Little Lost Man Creek at the bridge were electrofished to determine species present. Although no coho were present in Emerald and Bridge Creeks in either 1974 or 1975, coho were present in Bridge Creek in both 2004 and 2005. No coho were found in Lower Miller Creek in 2004 or 2005, there was one coho present in 1974. Tom McDonald Creek and Little Lost Man Creek at the bridge had coho present in 1974 but not in 1975.

Many other fish species were found to be common over all years of record (Appendix 6-1). In addition to coho salmon, steelhead salmon (*Oncorhynchus* mykiss) and cutthroat trout (*Oncorhynchus clarkii*) were commonly recorded. Coast range sculpin (*Cottus aleuticus*) and prickly sculpin (*Cottus asper*), three-spine stickleback (*Gasterosteus aculeatus*), Western brook lamprey (*Lampetra tridentata*) and several amphibians, including the Pacific giant salamander (*Dicamptodon tenebrosus*) and tailed frog (*Ascaphus truei*) were all species seen with some regularity in study tributaries. In 1974 and 1975 steelhead salmon were recorded at every site except Tom McDonald Creek, where they were only recorded in 1975. Coast range sculpin were recorded in every study tributary in 1974 and 1975 except Lower Miller and Bridge Creeks, where they were not recorded in 1974. Throughout all years of record, the Humboldt sucker *(Catostomus humboldtianus*) was only recorded at Harry Weir and Bridge Creeks in 1975.

To examine trends in growth rates, we compared length-weight regressions for the periods of record. Steeper slopes (greater than three) indicate that a fish is gaining weight at a faster unit of growth in length. The heavier a fish is for a given length, the higher its condition factor. A shift in the curves represents differences in the rate of growth. In the tributaries, the slopes of the regression lines ranged from 2.68 in 1975 in Bridge Creek to 3.24 in 2005 in Harry Weir Creek (Figures 6-1 – 6-5).

In Little Lost Man Creek at the bridge, a mostly pristine site, there was no significant difference in the slopes or intercepts between the 1974-75 and 2005 periods ($p = 0.48$ and 0.82, respectively). Nor was there a significant difference in slopes or intercepts between the 1974-75 and 1994 periods for Tom McDonald Creek ($p = 0.46$ and 0.69, respectively). Data for 2005 in Tom McDonald Creek are shown in Figure 6-3 for informational purposes, but 2005 data were not used in the regression analysis because there were only eight fish caught, all of which were in a smaller size class than the earlier

Table 6-1. Summary of coho salmon and trout presence/absence data collected via underwater snorkel observation during the summers of 2004 and 2005. X = fish present, 0 = fish absent and na = not snorkeled.

	2004		2005	
Site	Coho	Trout	Coho	Trout
Berry Glen Creek	O	X	O	X
Bond Creek	O	X	O	X
Bridge Creek	X	X	X	X
Cloquet Creek	O	X	X	X
Elam Creek	X	X	X	X
Fortyfour Creek	X	X	O	X
Godwood Creek	na	na	X	X
Harry Weir Creek	O	X	O	X
Hayes Creek	O	X	O	O
Larry Damm Creek	X	X	X	X
Little Lost Man Creek at bridge	X	X	X	X
Little Lost Man Creek at gage	X	X	X	X
Lost Man Creek below North Fork	X	X	X	X
Lost Man Creek above Larry Damm	X	X	X	X
McArthur Creek	X	X	X	X
Middle Fork Lost Man Creek	O	X	O	X
Miller Creek	O	X	O	X
North Fork Lost Man Creek	X	X	X	X
South Fork Lost Man Creek	O	X	O	X
Tom McDonald Creek	X	X	X	X
Upper Miller Creek	O	O	O	O
Upper Prairie Creek	X	X	X	X

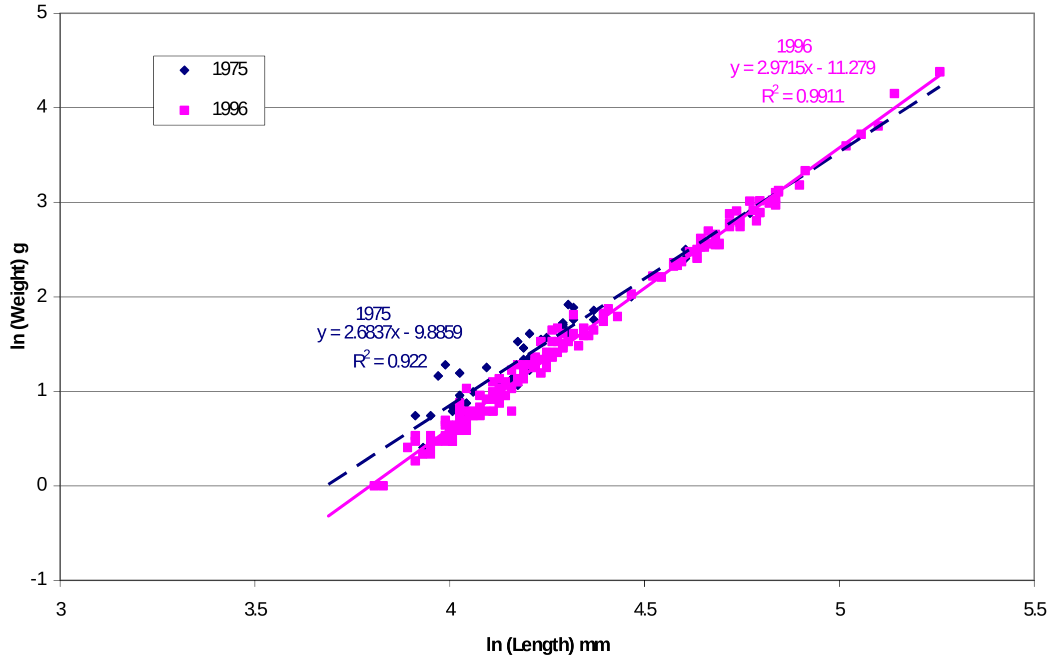


Figure 6-1. Length-weight comparisons for steelhead salmon collected from Bridge Creek**.**

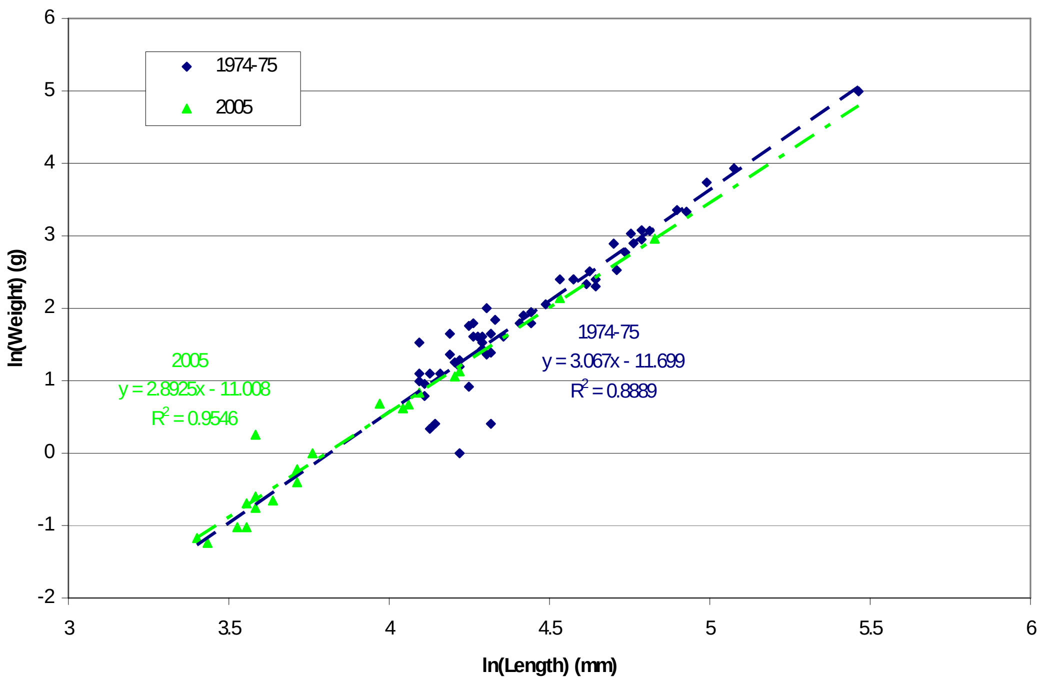


Figure 6-2. Length-weight comparisons for steelhead salmon collected from Little Lost Man Creek at the bridge**.**

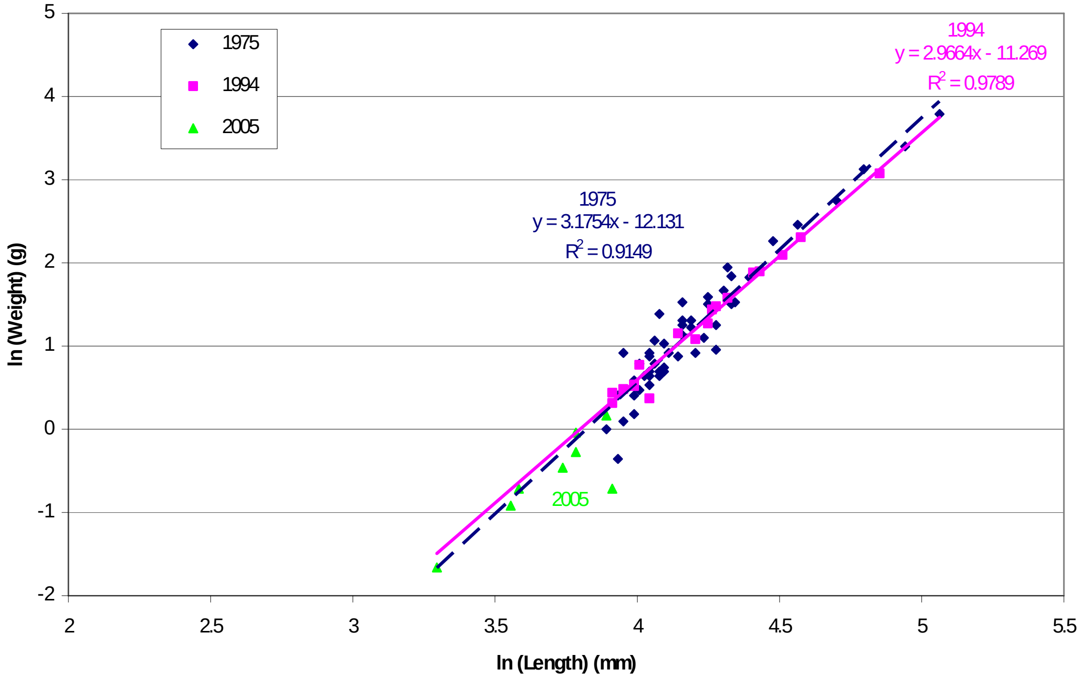


Figure 6-3. Length-weight comparisons for steelhead salmon collected from Tom McDonald Creek**.**

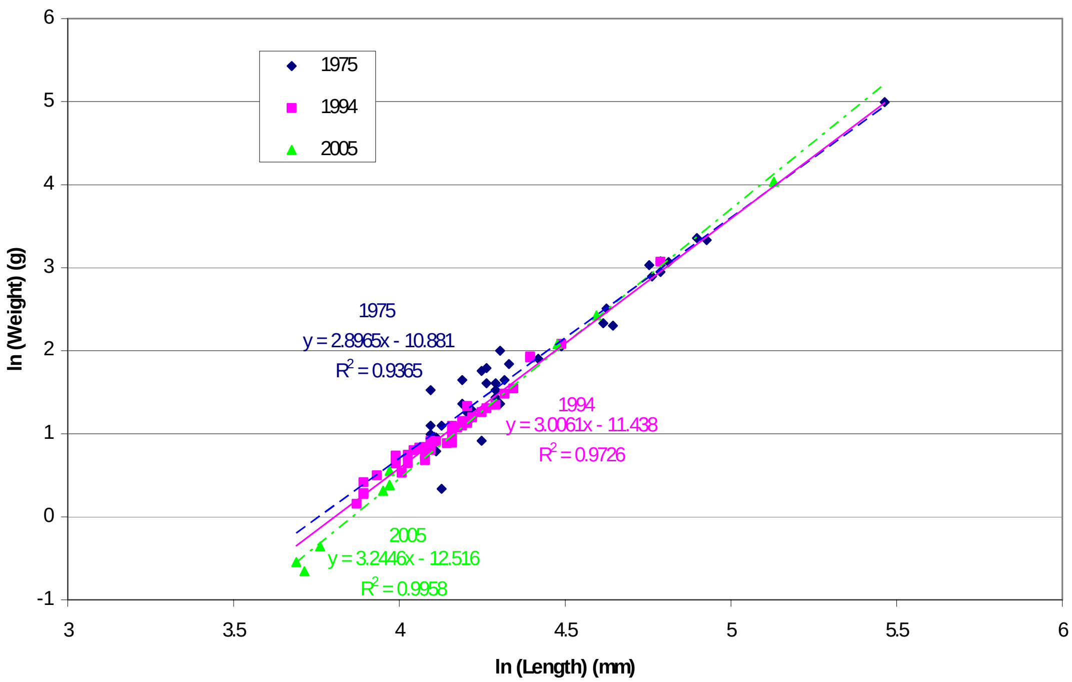


Figure 6-4. Length-weight comparisons for steelhead salmon collected from Harry Weir Creek**.**

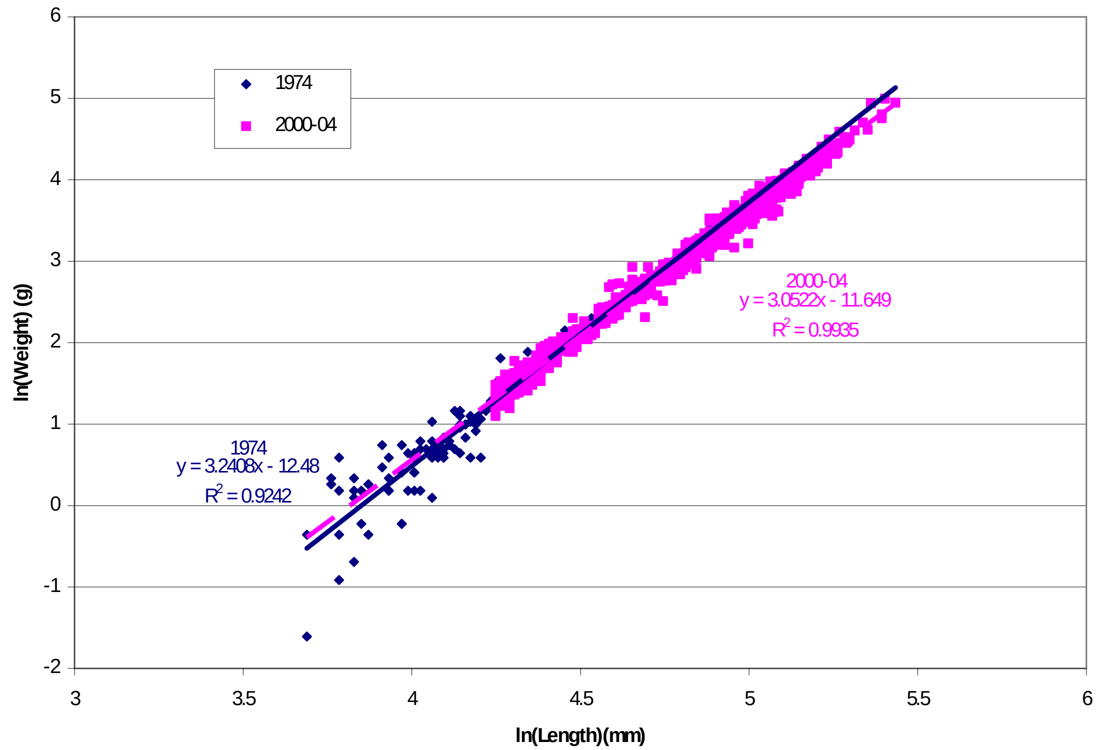


Figure 6-5. Length-weight comparisons for steelhead salmon collected from Redwood Creek near the Redwood Valley bridge**.**

data. In Harry Weir Creek, there was a significant increase in slopes through time at the 90 percent confidence interval ($p = 0.0835$), and a significant difference in intercepts ($p=0.0231$). Bridge Creek also displayed an increase in slopes through time, from 1975 to 1996 ($p = 0.0005$) and a significant change in intercepts ($p < 0.0001$). Only two steelhead were caught in 2005, so 2005 data were not included in the analysis.

The only mainstem station, Redwood Creek in Redwood Valley, is located about 84 km upstream of the mouth of Redwood Creek. This site is on private lands upstream of Redwood National Park boundaries and was not sampled by the USGS in 2005. There were no significant differences in slopes or intercepts between the 1974 and 2000 periods ($p = 0.53$ and $p = 0.31$, respectively).

Discussion and Conclusions

Sample sizes of fish captured or detected were small, so our results must be considered preliminary. For example, only one coho was found in Miller Creek in the 1970's, and none in the current study. Although coho were not found in Bridge Creek in 1974 or 1975, they were detected in this stream in 2004 and 2005. Our results suggest an improvement in steelhead fish condition in Harry Weir and Bridge Creeks over the last 30 years. No significant change was detected at either the pristine site (Little Lost Man Creek) or a site that has had continual timber harvest since the 1970's (Redwood Creek in Redwood Valley.

More fish data exist for the Redwood Creek basin than were used in the above analysis (Appendix 6-1), but it is difficult to compare sample results from different seasons collected through different techniques at different sites. In addition, 'fish condition' has been defined in various ways in the literature. Currently, fish trapping efforts at two sites on Redwood Creek by California Department of Fish and Game are generating additional data. As more data are collected, more advanced analyses of fish condition will be possible. Dr. Walt Duffy and Katherine McLaughlin at Humboldt State University are currently collecting more information on juvenile steelhead weights and densities in the Redwood Creek basin, which will shed further light on this issue.

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Appendices

Appendix 1-1. All parameters surveyed during the spring of 2004.

Site	Invertebrates-Rapid Bioassessment	Amphibians	Periphyton
Larry Damm	x	x	x
Little Lost Man at bridge	x	x	x
Lost Man below North Fork	x	x	x
Little Lost Man at gage	x	x	x
Fortyfour	x	x	x
Bond	x	x	
Lower Miller	x	x	x
Hayes	x		x
McArthur	x	x	
Upper Prairie	x	x	
Tom McDonald	x	x	x
Bridge	x	x	x
Harry Weir	x	x	x
Lost Man above Larry Damm	x		
Cloquet	x		
South Fork Lost Man	x		
Middle Fork Lost Man	x		
Elam	x		
North Fork Lost Man	x		
Upper Miller	x		
Berry Glen			
Godwood			

Appendix 1-2. All parameters surveyed during the summer of 2004.

Site	Invertebrates -Surber sampler	Invertebrates- Rapid Bioassessment	Periphyton	Snorkeling Survey	Cross-section	Discharge	Pebble Count	Longitudinal Profile	LWD Survey
Larry Damm	x	x	x	x	x	x	x	x	x
Little Lost Man at bridge	x	x	x	x	x	x	x		x
Lost Man below North Fork	x	x	x	x	x	x	x		x
Little Lost Man at gage	x	x	x	x	x	x	x		x
Fortyfour	x	x	x	x	x	x	x	x	x
Bond	x	x		x	x	x	x	x	x
Miller	x	x	x	x	x	x	x	x	x
Hayes				x	x		x	x	x
McArthur	x	x		x	x	x	x	x	x
Upper Prairie	x	x		x	x	x	x		x
Tom McDonald	x	x	x	x	x	x	x	x	x
Bridge	x	x	x	x	x	x	x		x
Harry Weir	x	x	x	x	x	x	x	x	x
Lost Man above Larry Damm	x	x		x	x	x	x	x	x
Cloquet	x	x		x	x	x	x	x	x
South Fork Lost Man	x	x		x	x	x	x	x	x
Middle Fork Lost Man	x	x		x	x	x	x	x	x
Elam	x	x		x	x	x	x	x	x

Appendix 1-2. continued.

Site	Invertebrates - Surber sampler	Invertebrates- Rapid Bioassessment	Periphyton	Snorkeling Survey	Cross-section	Discharge	Pebble Count	Longitudinal Profile	LWD Survey
North Fork Lost Man									
Upper Miller	x	x		x	x	x	x	x	x
Berry Glen					x		x	x	x
Godwood									

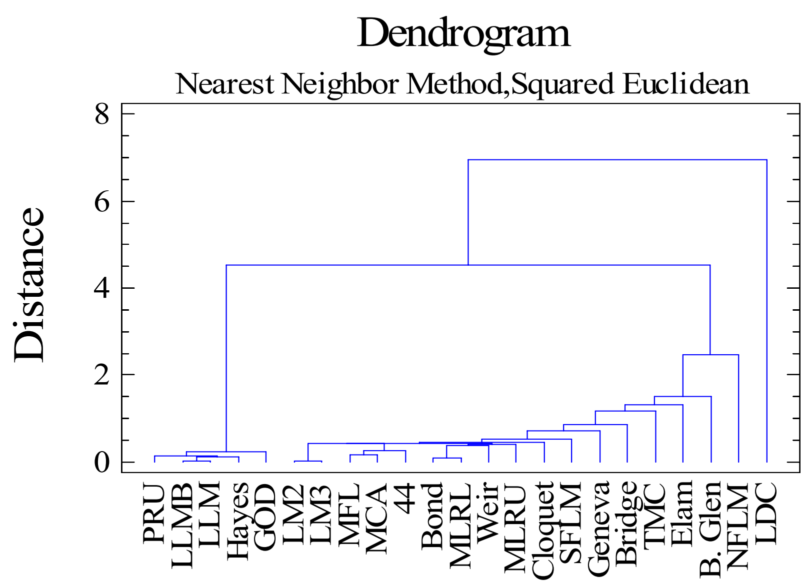
Appendix 1-3. All parameters surveyed during the spring of 2005.

Site	Invertebrates-Surber sampler	Invertebrates-Rapid Bioassessment	Amphibians	Periphyton
Larry Damm		x	x	x
Little Lost Man at bridge		x	x	x
Lost Man below North Fork		x	x	x
Little Lost Man at gage		x	x	x
Fortyfour		x	x	x
Bond		x	x	
Lower Miller		x	x	x
Hayes	x	x	x	x
McArthur		x	x	
Upper Prairie		x	x	
Tom McDonald		x	x	x
Bridge		x		x
Harry Weir		x	x	x
Lost Man above Larry Damm		x		
Cloquet		x		
South Fork Lost Man		x		
Middle Fork Lost Man		x		
Elam		x		
North Fork Lost Man		x		
Upper Miller		x		x
Berry Glen	x	x		
Godwood		x	x	x

Appendix 1-4. All parameters measured during the summer of 2005.

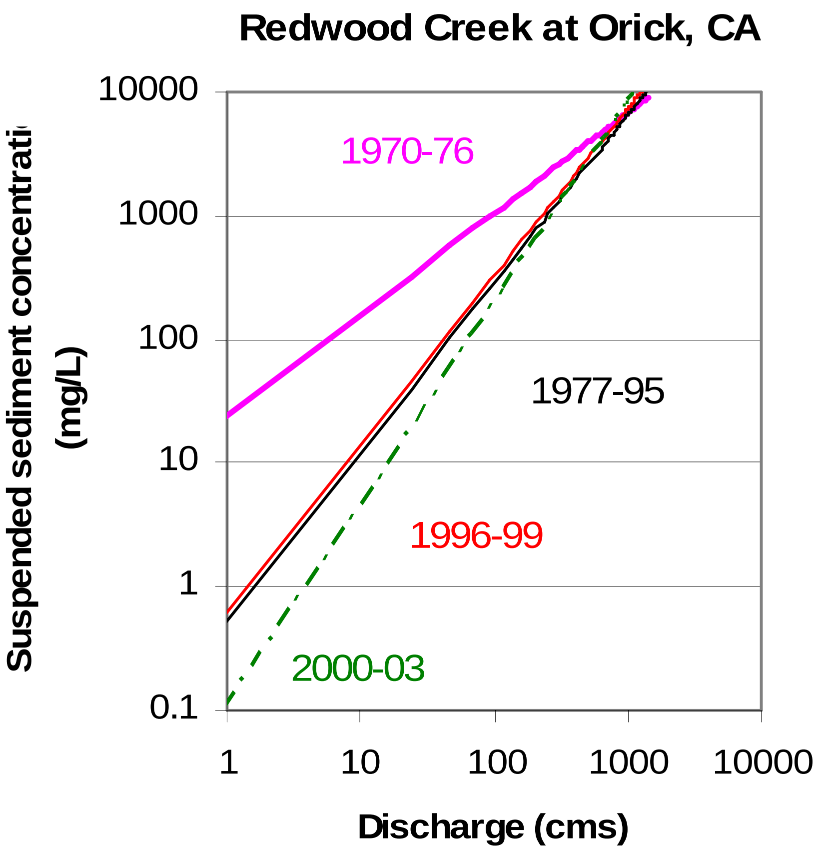
Site	Invertebrates-Rapid Bioassessment	Snorkeling Survery	Periphyton	S*
Larry Damm	x	x	x	x
Little Lost Man at bridge	x	x	x	x
Lost Man below North Fork	x	x	x	x
Little Lost Man at gage	x	x	x	x
Fortyfour	x	x	x	x
Bond	x	x		x
Miller	x	x	x	x
Hayes		x		
McArthur	x	x		x
Upper Prairie	x	x		x
Tom McDonald	x	x	x	x
Bridge	x	x	x	x
Harry Weir	x	x	x	x
Lost Man above Larry Damm	x	x		x
Cloquet	x	x		
South Fork Lost Man	x	x		
Middle Fork Lost Man	x	x		
Elam	x	x		x
North Fork Lost Man	x	x		x
Upper Miller	x	x	x	
Berry Glen		x		x
Godwood	x	x	x	x

Appendix 2-1. Cluster analysis of disturbance levels in sub-basins of Redwood Creek and Prairie Creek.



Cluster analysis separated sub-basins into general disturbance categories: Mostly pristine sites on the left (PRU-Upper Prairie, LLMB – Little Lost Man at Bridge, LLM – Little Lost Man at gage, Hayes, GOD- Godwood), Very high disturbance on the right (LDC – Larry Damm Creek), and many sub-basins in a moderate to high disturbance range (see Table 2-1 for specific basin information. Streams in the middle range are: LM2- Lost Man upstream of Larry Damm, LM3- Lost Man downstream of North Fork, MFL- Middle Fork Lost Man, MCA- McArthur, 44-Fortyfour, Bond, MLRL- Lower Miller, Weir- Harry Weir, MLRU – Upper Miller, Cloquet, SFLM- South Fork Lost Man, Geneva, Bridge, TMC- tom McDonald, Elam, B. Glen – Berry Glen, and NFLM- North Fork Lost Man.

Appendix 2-2. Suspended sediment discharge rating curves for Redwood Creek at Orick.



Appendix 4-1. Percentages of functional groups calculated for benthic macroinvertebrates sampled from tributaries of Redwood Creek with a 250 μm Surber sampler. Omnivores not included in calculations.

Site	Sampling Date	Filtering-Collector	Gathering-Collector	Scraper	Shredder	Predator	Piercer-Herbivore	Individuals per square meter
Upper Prairie	9/10/2004	6.7	54.0	20.4	2.8	16.0	0.0	2000
Hayes	5/15/1974	0.5	60.8	0.9	17.7	20.1	0.0	1200
	9/14/1974	0.1	56.6	12.8	15.3	15.2	0.0	14000
	5/29/1975	3.8	42.0	34.1	2.9	17.2	0.0	5000
	9/20/1975	0.1	64.0	3.6	13.1	19.2	0.0	9200
	6/17/2004	4.0	27.3	10.2	8.2	50.3	0.0	470
	5/24/2005	5.7	56.9	7.5	4.7	24.5	0.0	210
Little Lost Man at gage	5/10/1974	7.8	38.2	11.6	2.2	40.1	0.0	4100
	9/14/1974	0.5	47.2	23.7	2.9	24.7	0.9	15000
	6/2/1975	0.2	52.3	6.1	17.4	23.9	0.0	6900
	9/20/1975	1.7	55.6	18.3	6.7	16.8	0.8	26000
	5/27/2004	0.1	56.0	4.1	10.0	21.5		1380
	9/13/2004	1.7	48.3	21.2	3.3	25.5		2106
Little Lost Man at bridge	5/10/1974	0.5	51.4	20.0	5.9	22.2	0.0	4200
	6/3/2004	1.6	48.5	7.4	10.8	31.8	0.0	1600
	9/13/2004	1.0	59.0	14.4	4.1	21.6	0.0	1700
Elam	5/15/1974	1.1	37.5	5.1	33.0	23.3	0.0	1000
	9/21/1974	8.6	31.0	0.6	32.4	27.2	0.0	1300
	9/15/2004	2.2	78.9	3.0	7.3	8.6	0.0	2000
Berry Glen	5/12/2005	1.3	61.9	6.6	4.7	23.8	0.0	350

Appendix 4-1. continued.

Site	Sampling Date	Filtering-Collector	Gathering-Collector	Scraper	Shredder	Predator	Piercer-Herbivore	Individuals per square meter
Cloquet	5/14/1974	13.4	50.8	4.6	4.2	26.9	0.0	3500
	9/21/1974	2.2	69.6	4.1	13.5	10.5	0.0	10000
	9/16/2004	0.0	55.6	11.8	6.6	26.1	0.0	1400
Harry Weir	5/13/1974	9.0	16.9	11.9	6.8	55.4	0.0	1800
	9/16/1974	0.3	67.6	6.4	9.3	16.4	0.1	3800
	6/1/1975	1.0	59.2	9.4	5.8	24.5	0.0	1100
	9/16/1975	0.0	52.8	10.1	2.7	34.4	0.0	1000
	6/22/2004	1.1	49.4	8.0	14.6	26.9	0.0	390
	8/31/2004	1.8	49.6	7.5	5.5	35.6	0.0	1000
Bond	9/16/2004	2.0	54.7	19.8	9.6	14.0	0.0	540
Lower Miller	5/14/1974	2.7	23.3	20.7	8.6	44.8	0.0	580
	9/17/1974	10.3	29.3	15.6	2.0	42.4	0.5	2300
	5/31/1975	4.1	19.2	18.4	9.2	49.1	0.0	1200
	9/21/1975	15.2	69.2	1.5	7.0	7.1	0.0	4500
	6/15/2004	0.5	37.1	4.9	22.1	35.4	0.0	260
	9/16/2004	0.6	62.9	10.8	2.2	23.5	0.0	460
South Fork Lost Man	9/10/04	4.8	26.6	29.4	6.9	32.3	0.0	1100

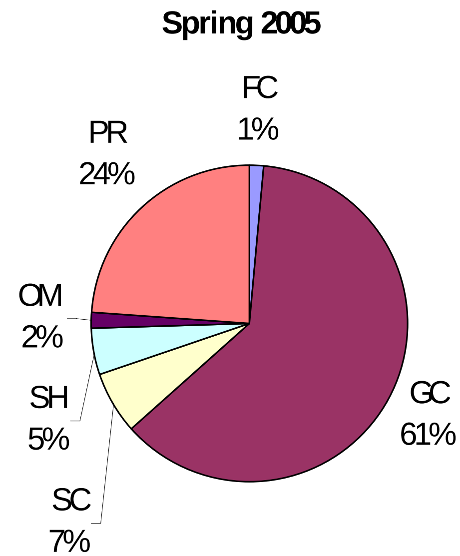
Appendix 4-1. continued.

Site	Sampling Date	Filtering-Collector	Gathering-Collector	Scraper	Shredder	Predator	Piercer-Herbivore	Individuals per square meter
Lost Man below North Fork	5/10/1974	1.7	56.1	16.7	5.4	20.1	0.0	1000
	9/15/1974	3.8	37.8	29.7	4.1	24.6	0.0	7000
	6/2/1975	1.7	8.0	32.2	0.7	57.5	0.0	4100
	9/17/1975	7.4	29.6	36.4	8.2	18.4	0.0	6300
	6/2/2004	0.8	58.2	6.0	5.7	29.2	0.0	1200
	9/13/2004	3.3	59.3	13.5	4.9	19.1	0.0	3200
Upper Miller	5/10/1974	0.3	80.5	0.8	12.7	5.8	0.0	4400
	9/17/1974	0.8	61.6	7.9	18.4	11.3	0.0	5500
	5/31/1975	0.2	49.0	2.2	45.1	3.5	0.0	6000
	9/21/1975	1.0	58.7	0.4	0.3	39.5	0.0	13000
	9/9/2004	0.0	43.0	12.1	12.9	32.0	0.0	270
Middle Fork Lost Man	9/10/2004	8.8	29.5	37.7	6.0	18.1	0.0	630
Fortyfour	6/14/2004	0.3	44.9	11.6	20.0	23.1	0.0	230
	9/16/2004	3.3	45.1	19.8	14.1	17.7	0.0	270
McArthur	9/15/2004	1.3	59.8	14.9	8.7	15.3	0.0	2300
Tom McDonald	5/14/1974	7.8	59.2	1.3	6.7	25.0	0.0	1200
	9/24/1974	1.7	61.5	6.7	6.8	22.8	0.5	9300
	5/30/1975	0.0	57.6	9.8	11.7	21.0	0.0	600
	9/16/1975	4.0	72.6	7.0	11.1	5.3	0.0	8200
	6/23/2004	0.5	65.4	4.3	6.3	23.5	0.0	1300
	8/31/2004	10.5	62.2	9.2	2.6	15.5	0.0	4500

Appendix 4-1. continued.

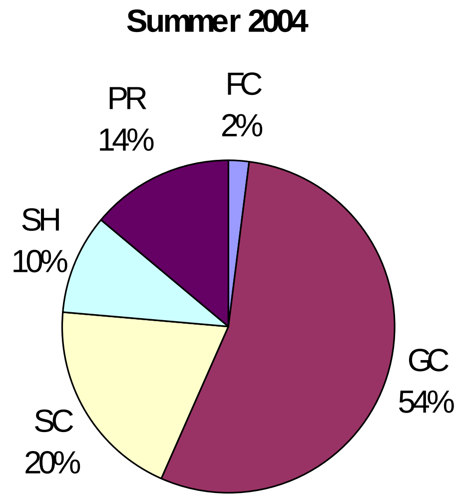
Site	Sampling Date	Filtering-Collector	Gathering-Collector	Scraper	Shredder	Predator	Piercer-Herbivore	Individuals per square meter
Bridge	5/13/1974	1.3	76.7	1.0	0.7	20.2	0.0	1100
	9/16/1974	1.5	55.2	18.3	1.3	14.7	8.9	9500
	6/1/1975	3.4	86.1	1.1	0.4	9.0	0.0	1400
	9/16/1975	0.0	0.0	50.0	0.0	50.0	0.0	22
	6/22/2004	1.0	69.3	14.4	5.5	9.7	0.0	2900
	9/9/2004	9.1	43.1	11.9	1.5	34.4	0.0	1400
North Fork Lost Man	9/13/2004	0.6	62.9	10.8	2.2	23.5	0.0	1700
Larry Damm	5/26/2004	0.0	52.8	12.3	10.8	24.1	0.0	1200
	8/30/2004	0.9	48.2	25.5	4.2	21.2	0.0	1200

Berry Glen Creek



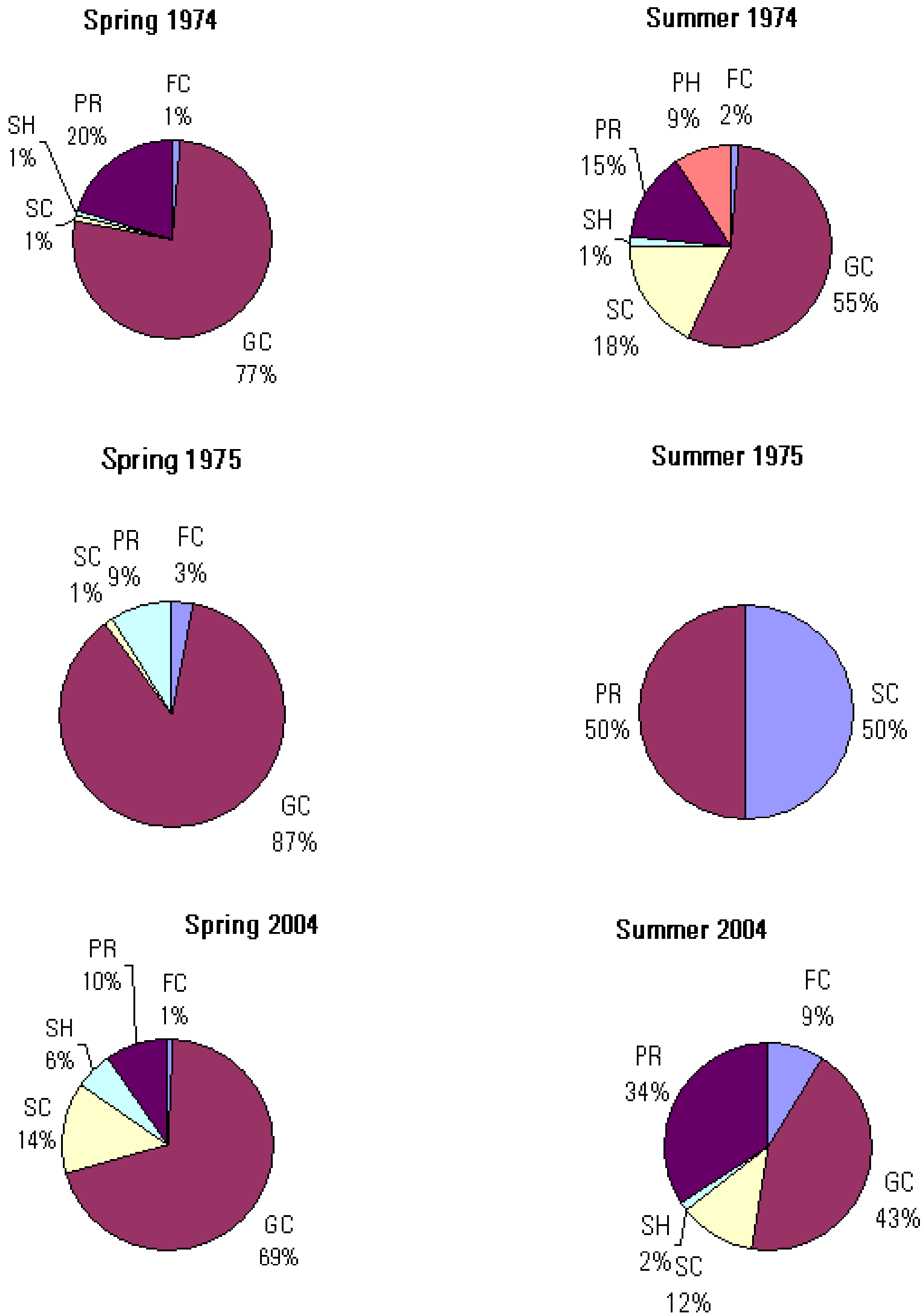
Appendix 4-2a. Functional feeding group percentages for Berry Glen Creek. GC=gathering collector, FC=filtering collector, SC=scraper, SH=shredder, PR=predator, PH=piercer herbivore.

Bond Creek



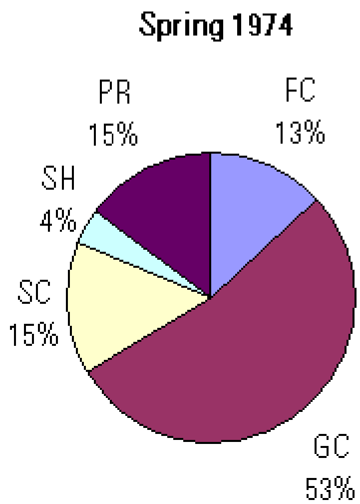
Appendix 4-2b. Functional feeding group percentages for Bond Creek. GC=gathering collector, FC=filtering collector, SC=scraper, SH=shredder, PR=predator, PH=piercer herbivore.

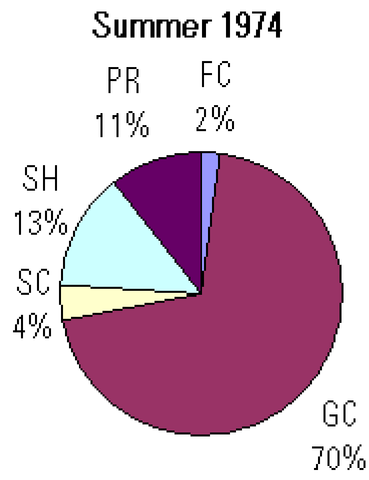
Bridge Creek

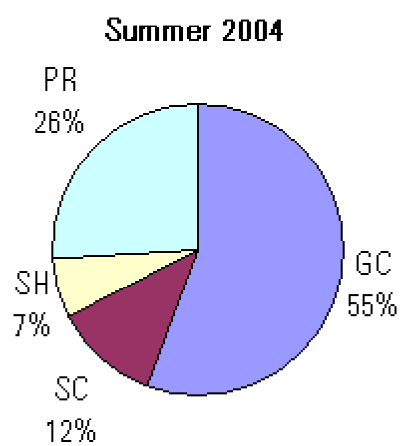


Appendix 4-2c. Functional feeding group percentages for Bridge Creek. GC=gathering collector, FC=filtering collector, SC=scraper, SH=shredder, PR=predator, PH=piercer herbivore.

Cloquet Creek

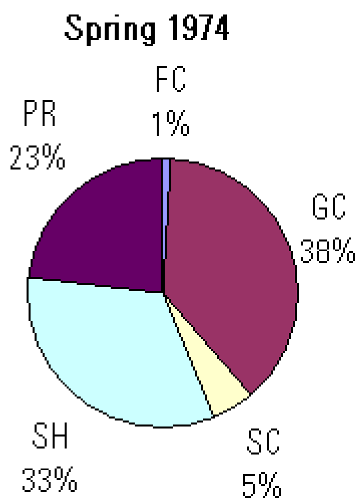


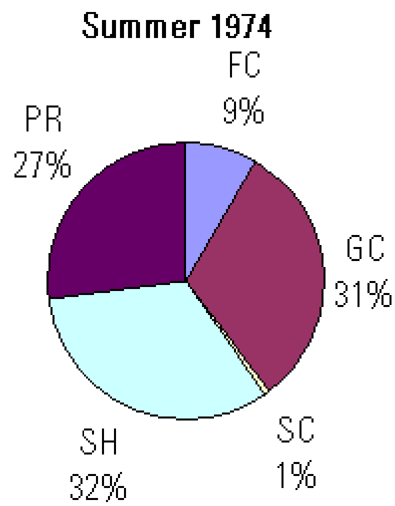


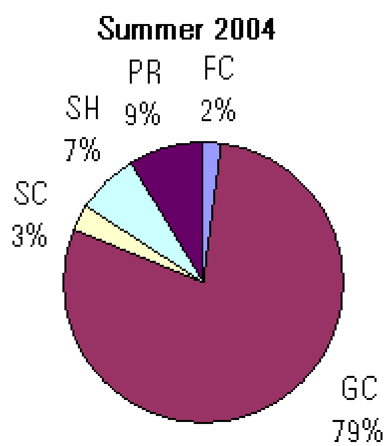


Appendix 4-2d. Functional feeding group percentages for Cloquet Creek. GC=gathering collector, FC=filtering collector, SC=scraper, SH=shredder, PR=predator, PH=piercer herbivore.

Elam Creek

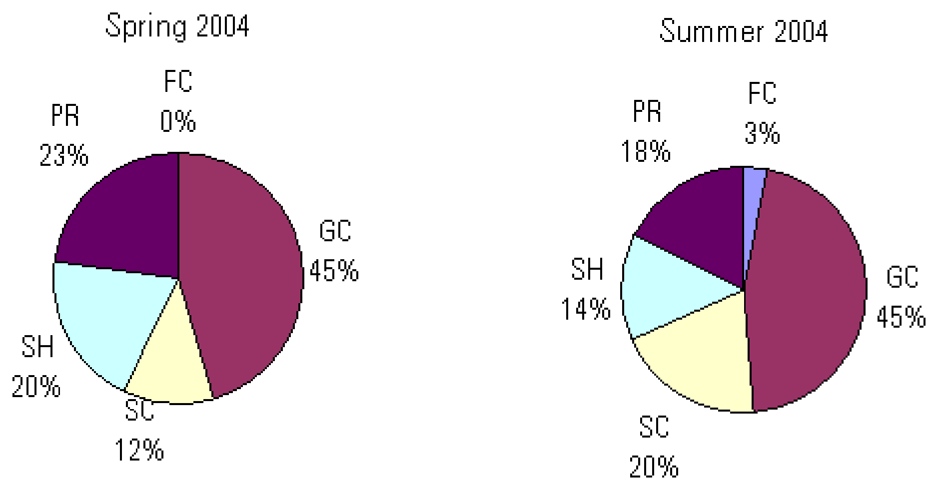






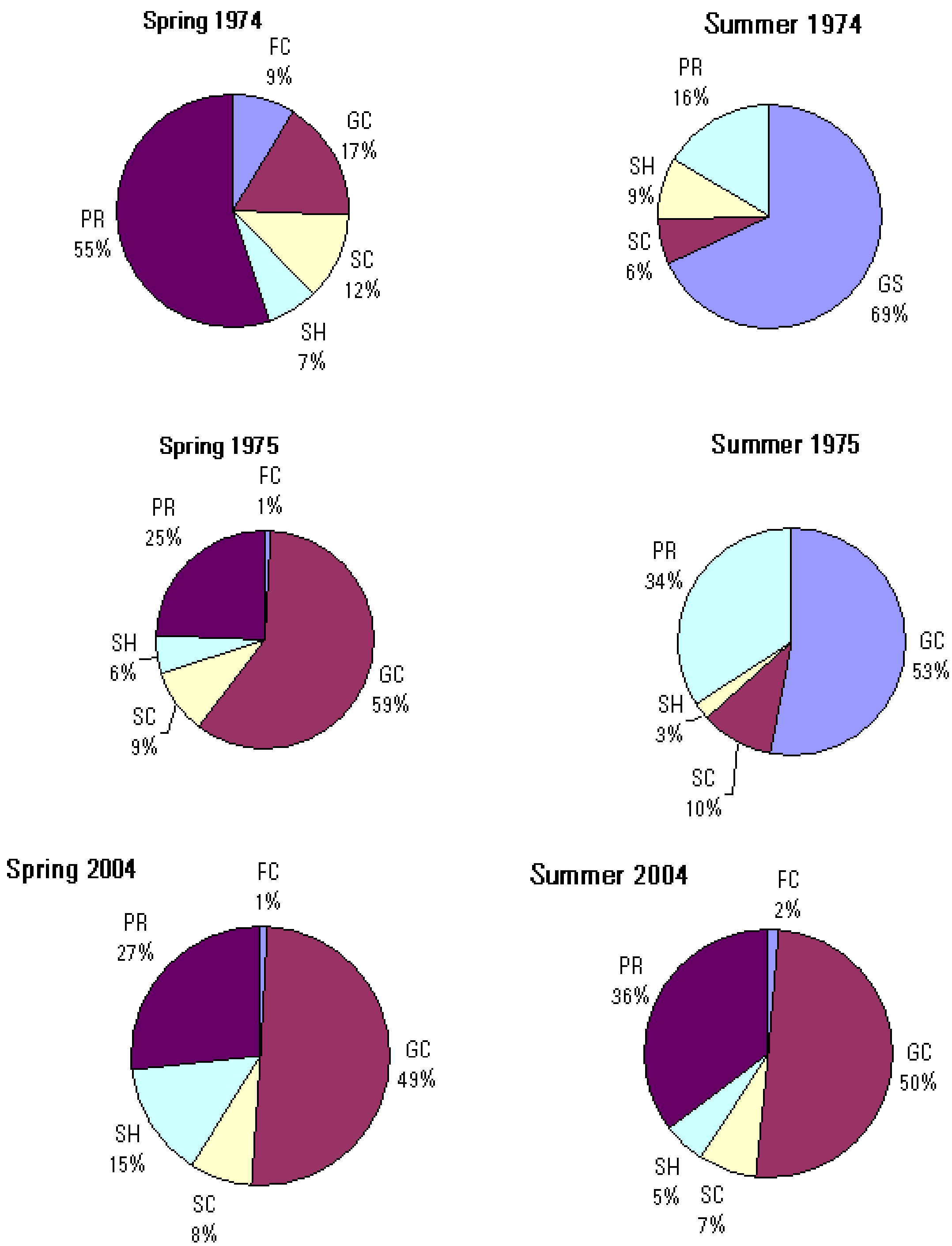
Appendix 4-2e. Functional feeding group percentages for Elam Creek. GC=gathering collector, FC=filtering collector, SC=scraper, SH=shredder, PR=predator, PH=piercer herbivore.

Fortyfour Creek



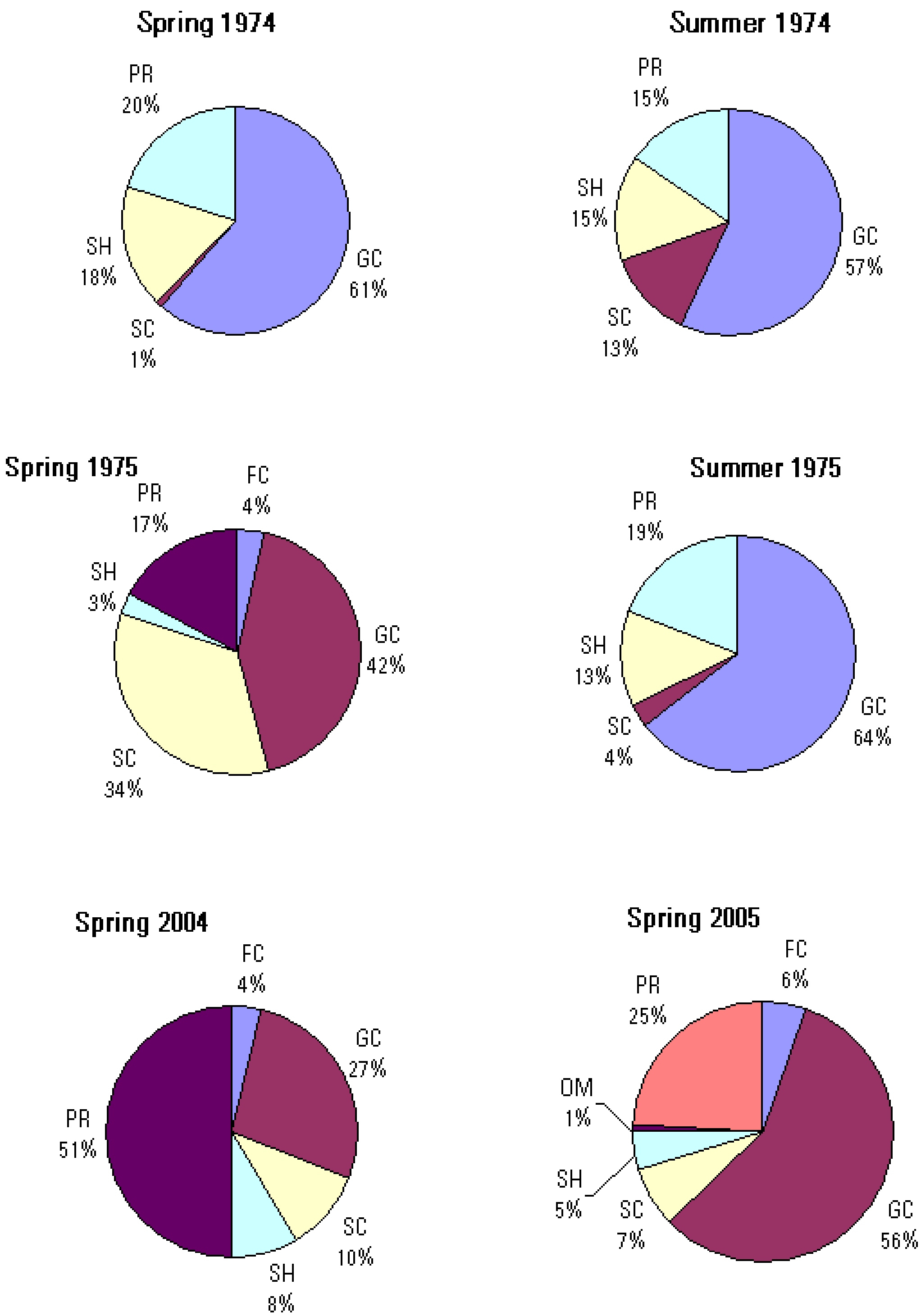
Appendix 4-2f. Functional feeding group percentages for Fortyfour Creek. GC=gathering collector, FC=filtering collector, SC=scraper, SH=shredder, PR=predator, PH=piercer herbivore.

Harry Weir Creek



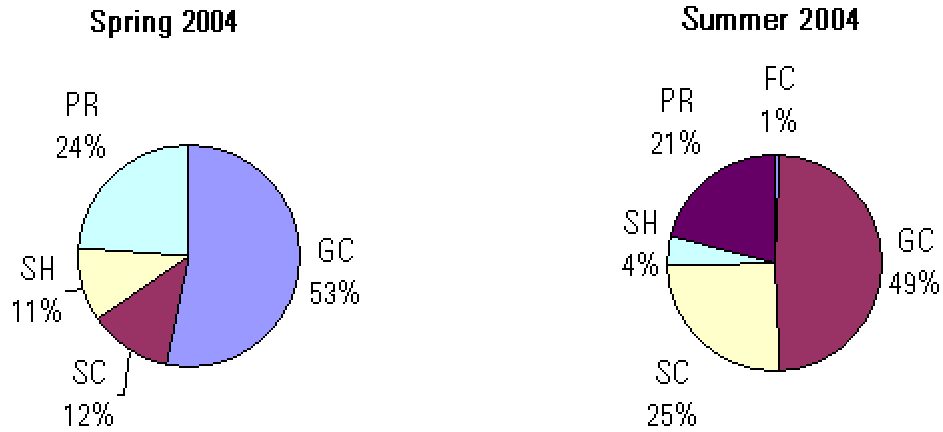
Appendix 4-2g. Functional feeding group percentages for Harry Weir Creek. GC=gathering collector, FC=filtering collector, SC=scraper, SH=shredder, PR=predator, PH=piercer herbivore.

Hayes Creek



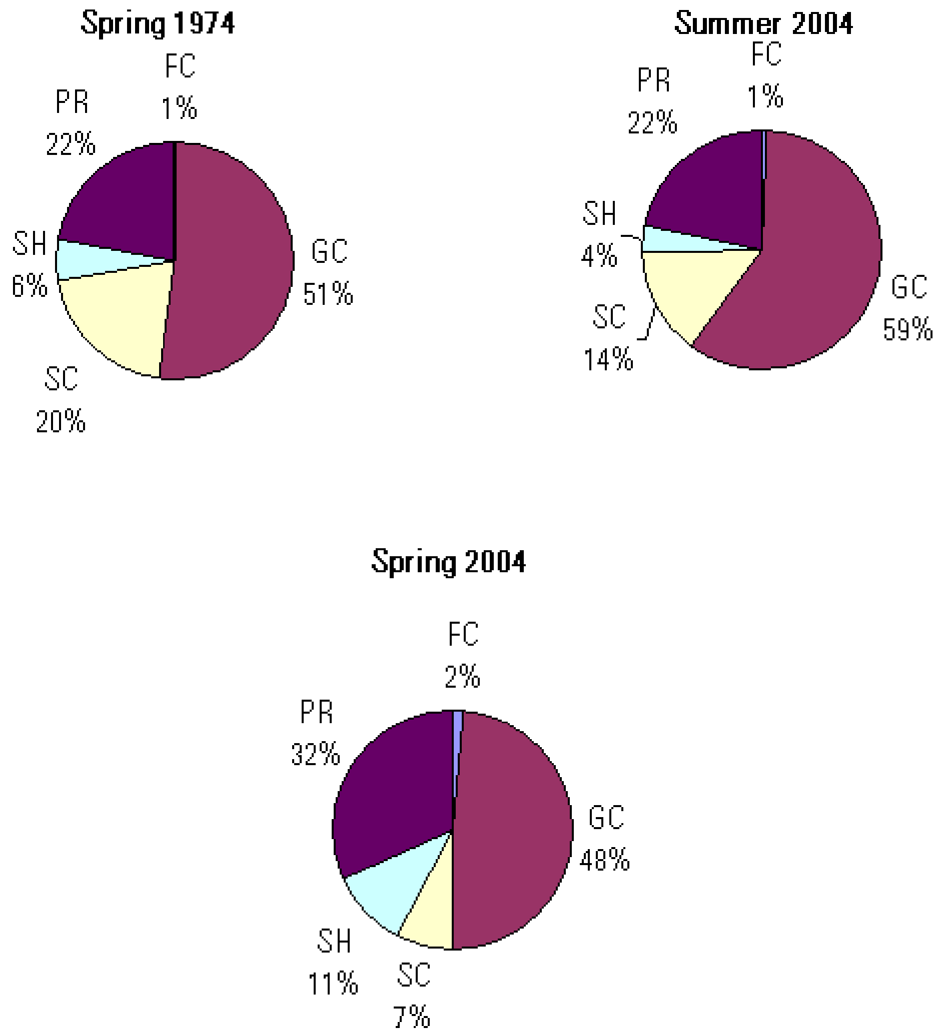
Appendix 4-2h. Functional feeding group percentages for Hayes Creek. GC=gathering collector, FC=filtering collector, SC=scraper, SH=shredder, PR=predator, PH=piercer herbivore.

Larry Damm Creek



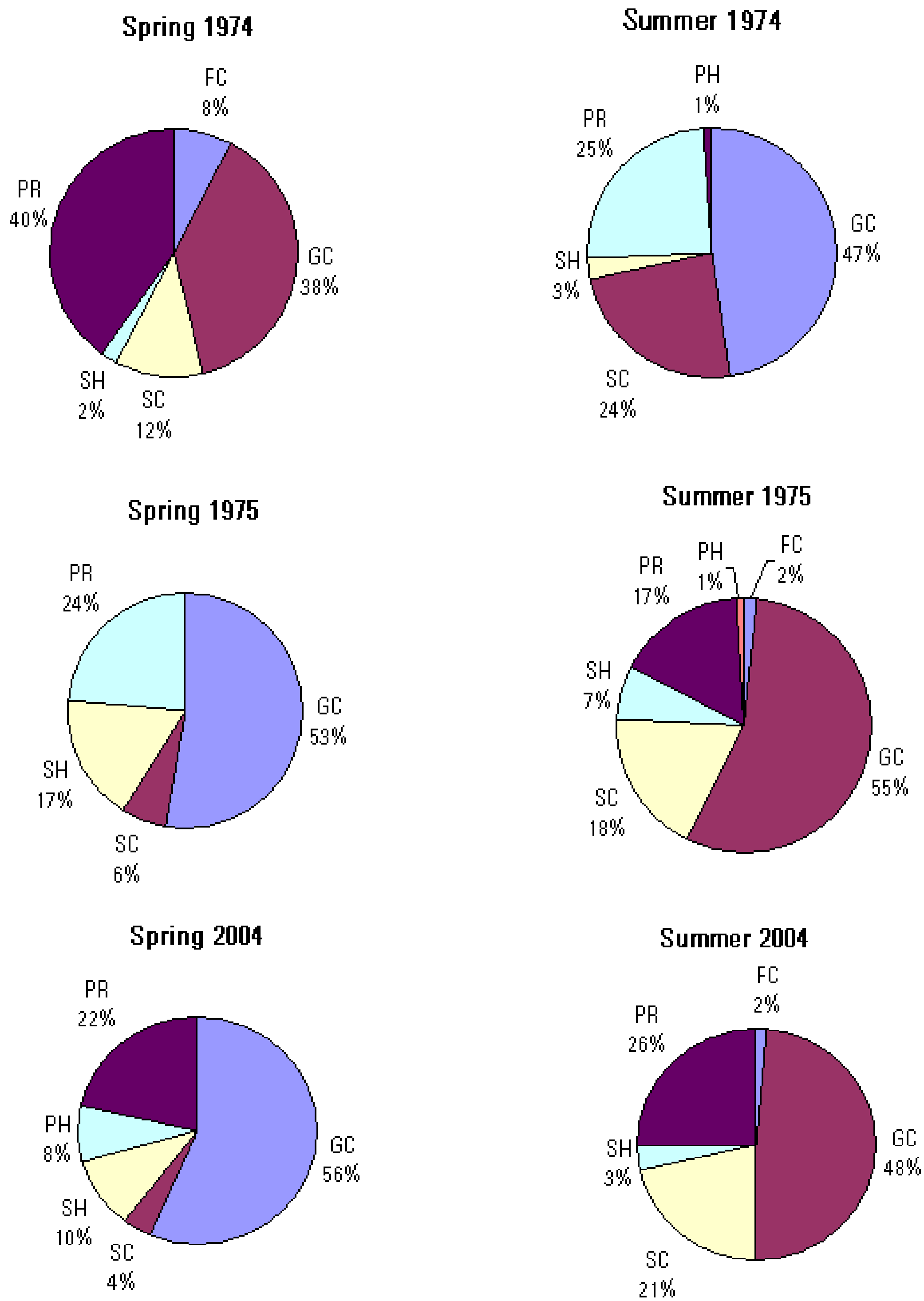
Appendix 4-2i. Functional feeding group percentages for Larry Damm Creek. GC=gathering collector, FC=filtering collector, SC=scraper, SH=shredder, PR=predator, PH=piercer herbivore.

Little Lost Man Creek at the bridge



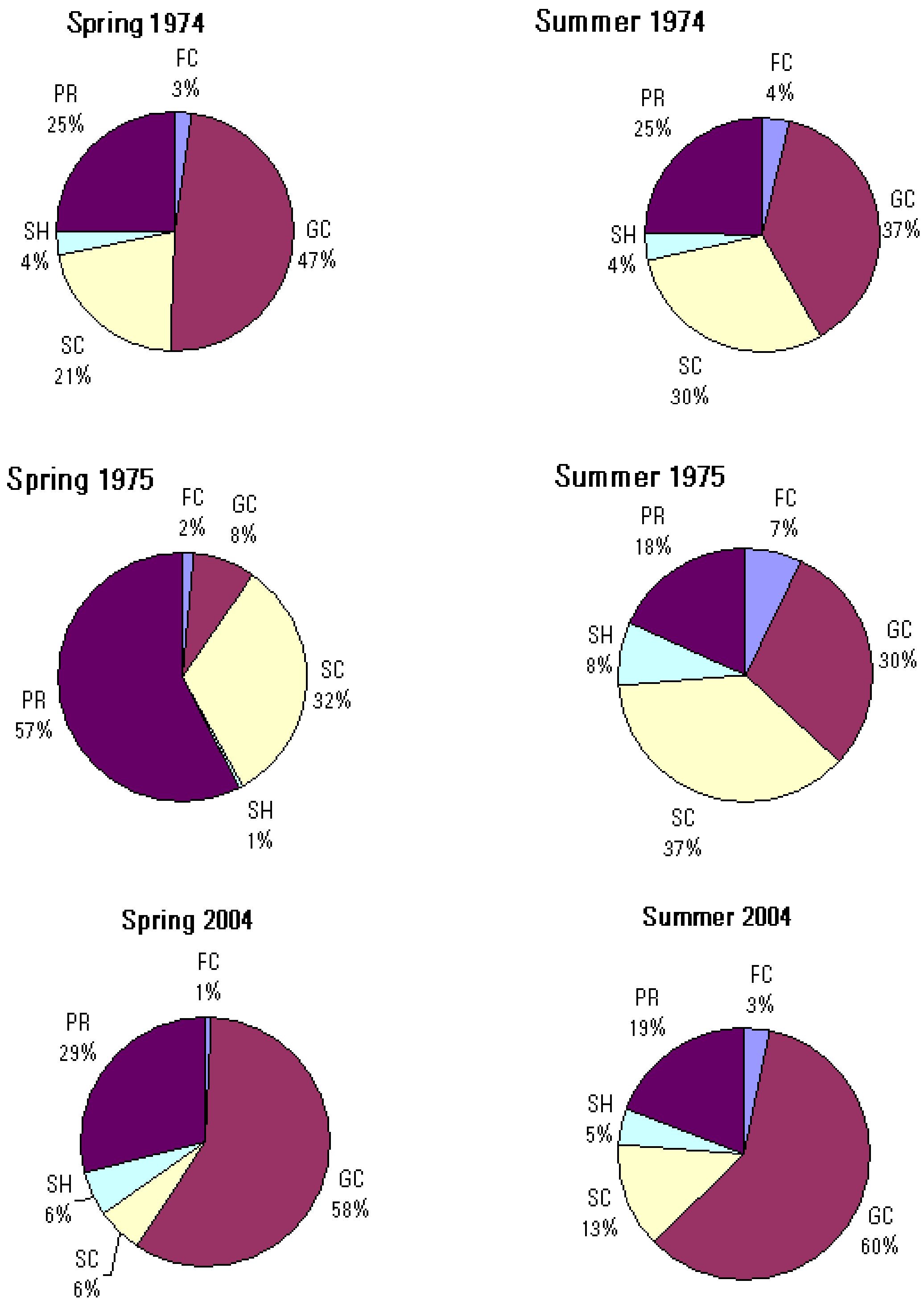
Appendix 4-2j. Functional feeding group percentages for Little Lost Man Creek at the bridge. GC=gathering collector, FC=filtering collector, SC=scraper, SH=shredder, PR=predator, PH=piercer herbivore.

Little Lost Man Creek at the gage



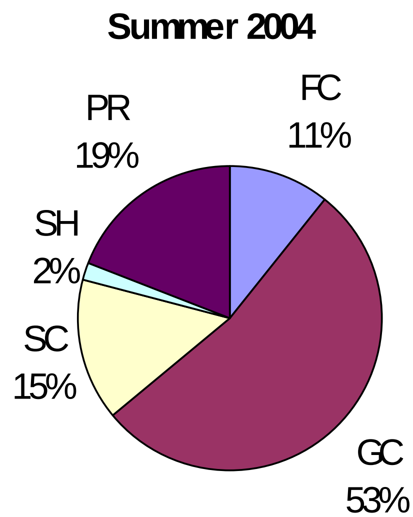
Appendix 4-2k. Functional feeding group percentages for Little Lost Man Creek at the gage. GC=gathering collector, FC=filtering collector, SC=scraper, SH=shredder, PR=predator, PH=piercer herbivore.

Lost Man Creek below North Fork Lost Man



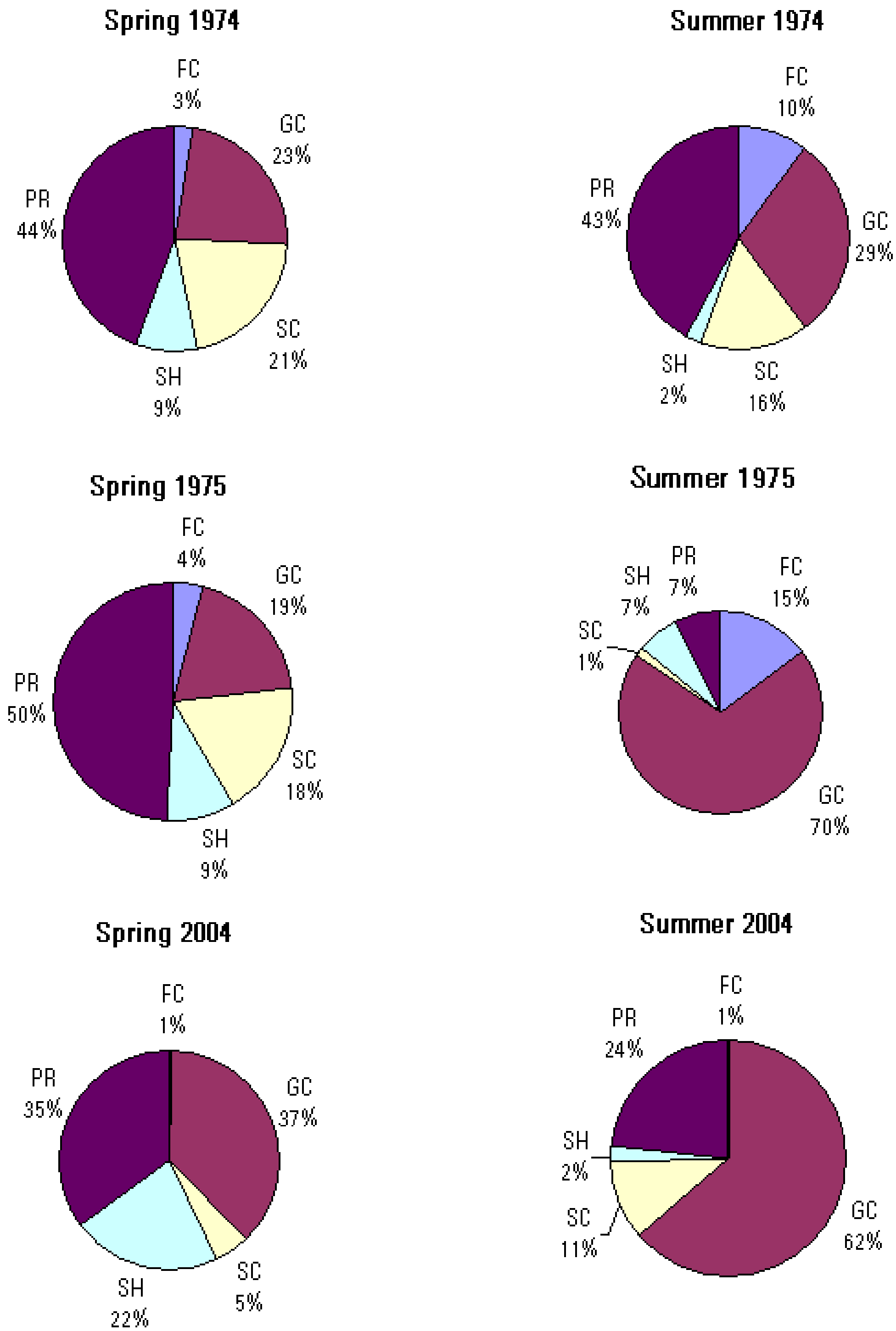
Appendix 4-2l. Functional feeding group percentages for Lost Man Creek below North Fork Lost Man. GC=gathering collector, FC=filtering collector, SC=scraper, SH=shredder, PR=predator, PH=piercer herbivore.

Lost Man Creek above Larry Damm Creek



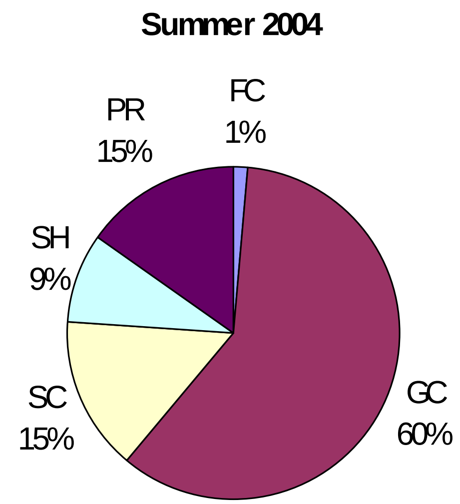
Appendix 4-2m. Functional feeding group percentages for Lost Man Creek above Larry Damm Creek. GC=gathering collector, FC=filtering collector, SC=scraper, SH=shredder, PR=predator, PH=piercer herbivore.

Lower Miller Creek



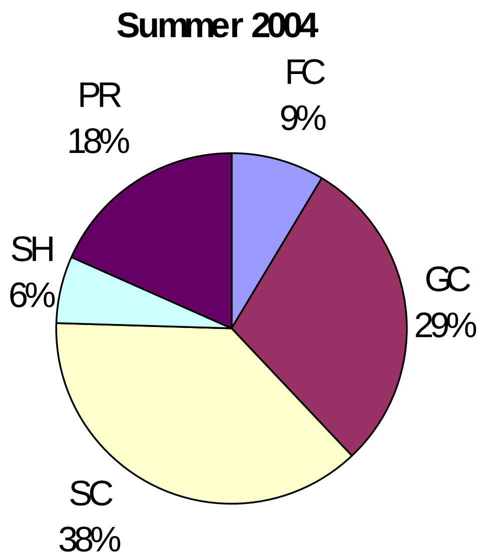
Appendix 4-2n. Functional feeding group percentages for Lower Miller Creek. GC=gathering collector, FC=filtering collector, SC=scraper, SH=shredder, PR=predator, PH=piercer herbivore.

McArthur Creek



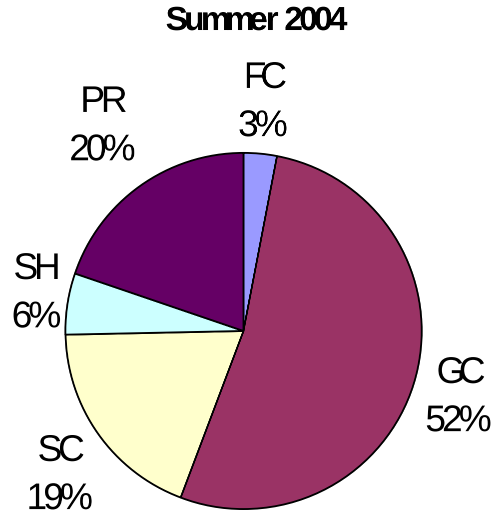
Appendix 4-2o. Functional feeding group percentages for McArthur Creek. GC=gathering collector, FC=filtering collector, SC=scraper, SH=shredder, PR=predator, PH=piercer herbivore.

Middle Fork Lost Man Creek



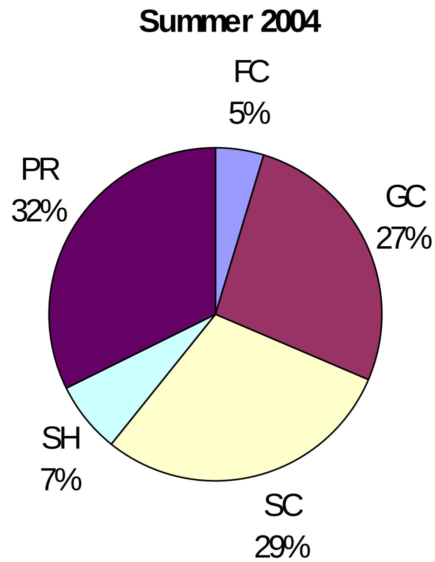
Appendix 4-2p. Functional feeding group percentages for Middle Fork Lost Man Creek. GC=gathering collector, FC=filtering collector, SC=scraper, SH=shredder, PR=predator, PH=piercer herbivore.

North Fork Lost Man Creek



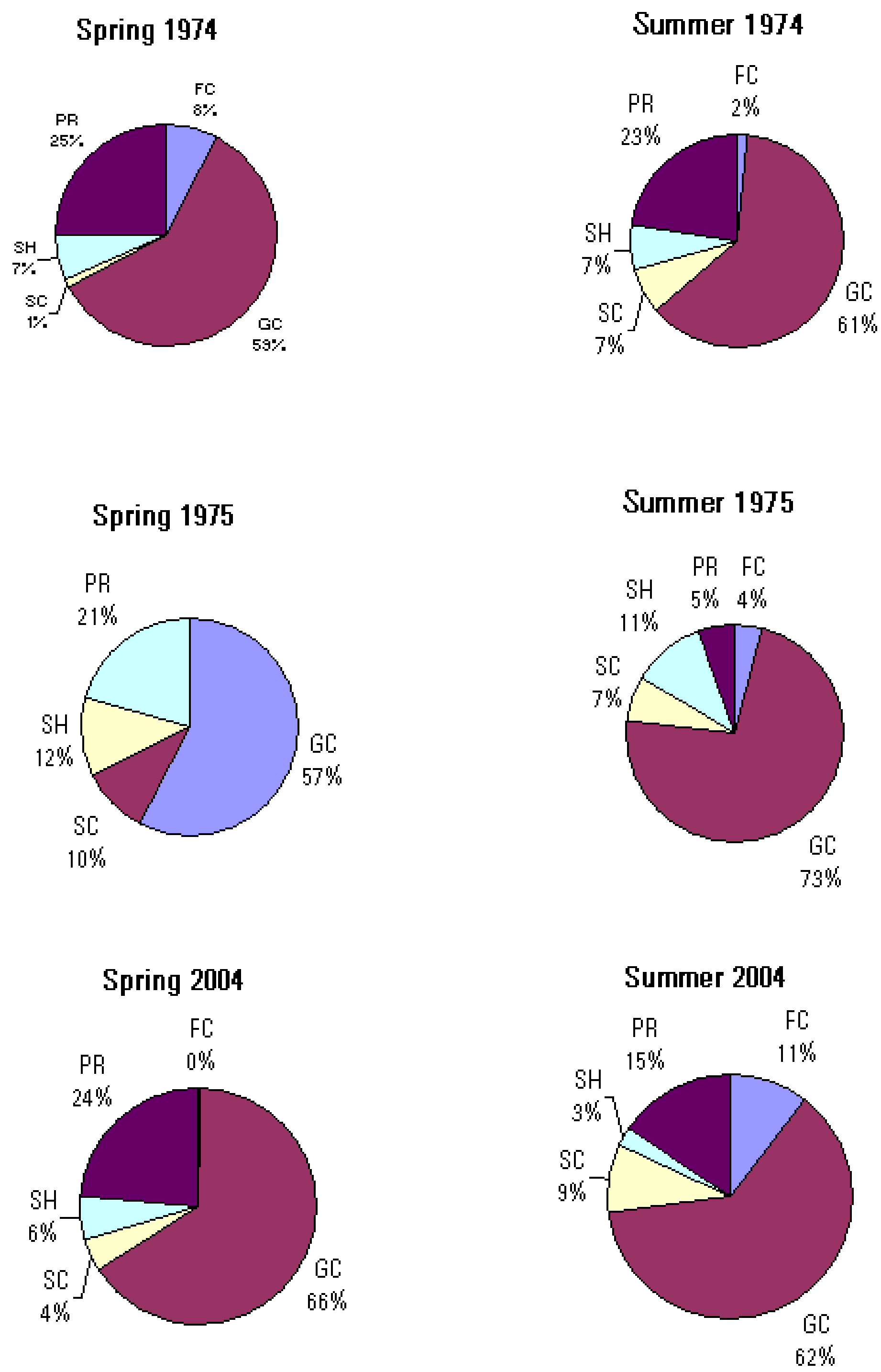
Appendix 4-2q. Functional feeding group percentages for North Fork Lost Man Creek. GC=gathering collector, FC=filtering collector, SC=scraper, SH=shredder, PR=predator, PH=piercer herbivore.

South Fork Lost Man Creek



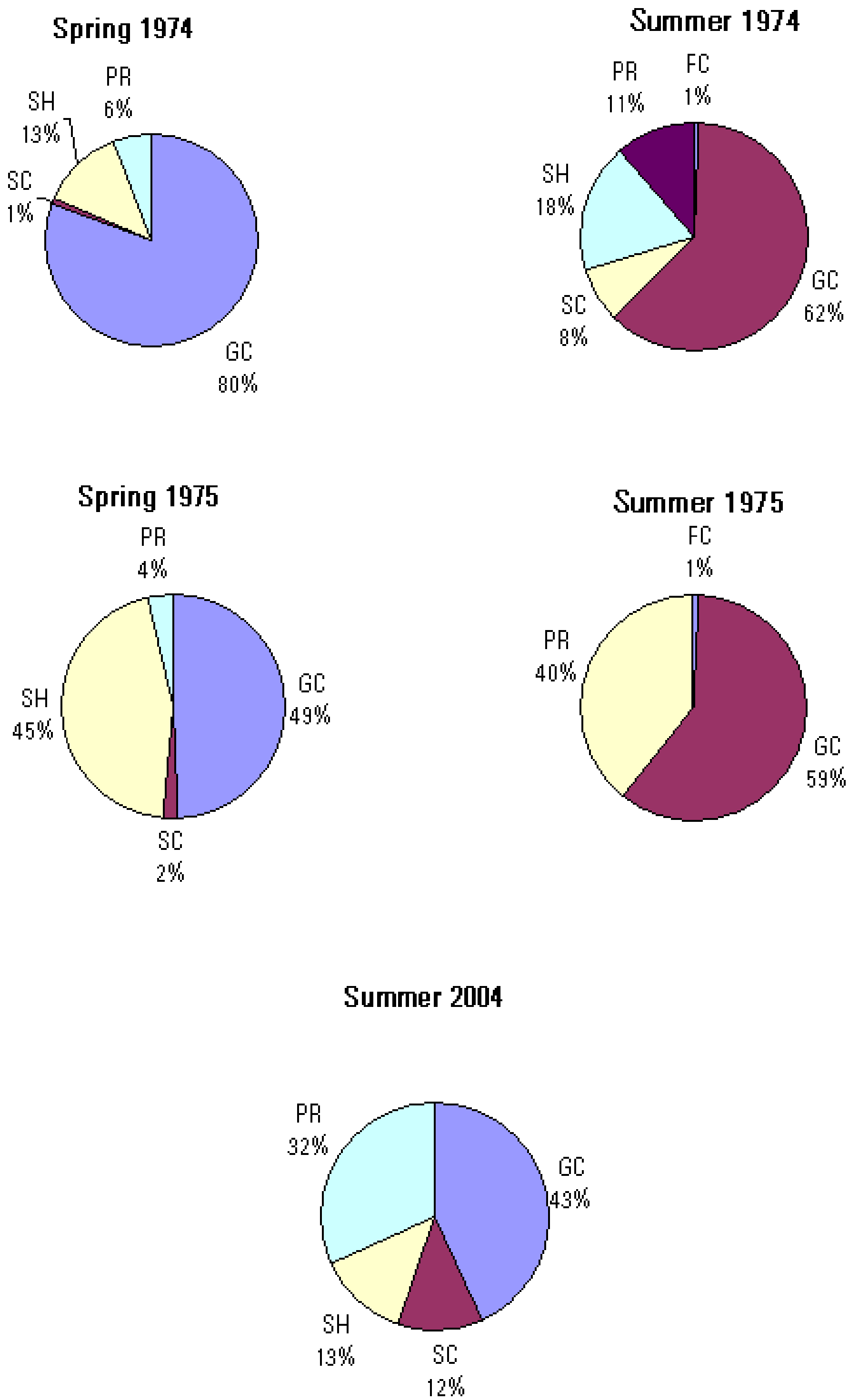
Appendix 4-2r. Functional feeding group percentages for South Fork Lost Man Creek. GC=gathering collector, FC=filtering collector, SC=scraper, SH=shredder, PR=predator, PH=piercer herbivore.

Tom McDonald Creek



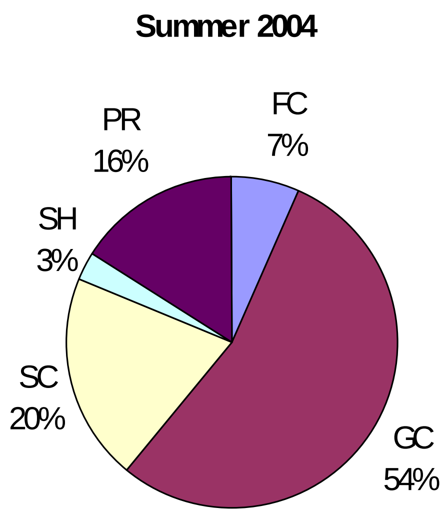
Appendix 4-2s. Functional feeding group percentages for Tom McDonald Creek. GC=gathering collector, FC=filtering collector, SC=scraper, SH=shredder, PR=predator, PH=piercer herbivore.

Upper Miller Creek

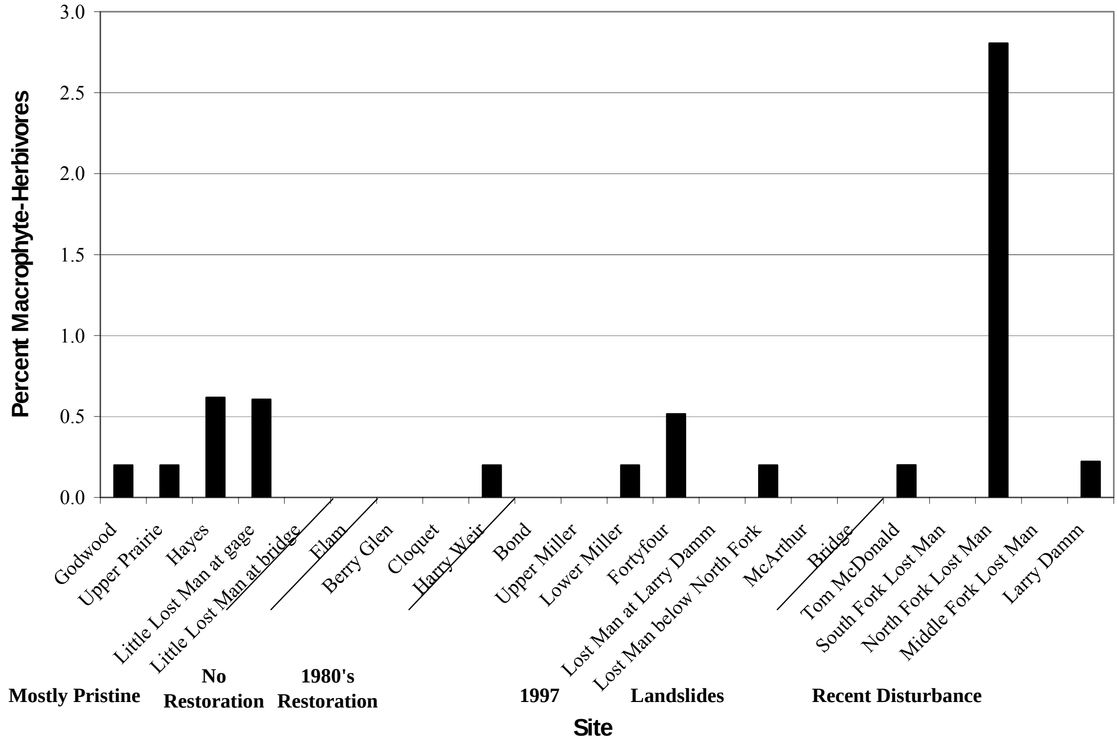


Appendix 4-2t. Functional feeding group percentages for Upper Miller Creek. GC=gathering collector, FC=filtering collector, SC=scraper, SH=shredder, PR=predator, PH=piercer herbivore.

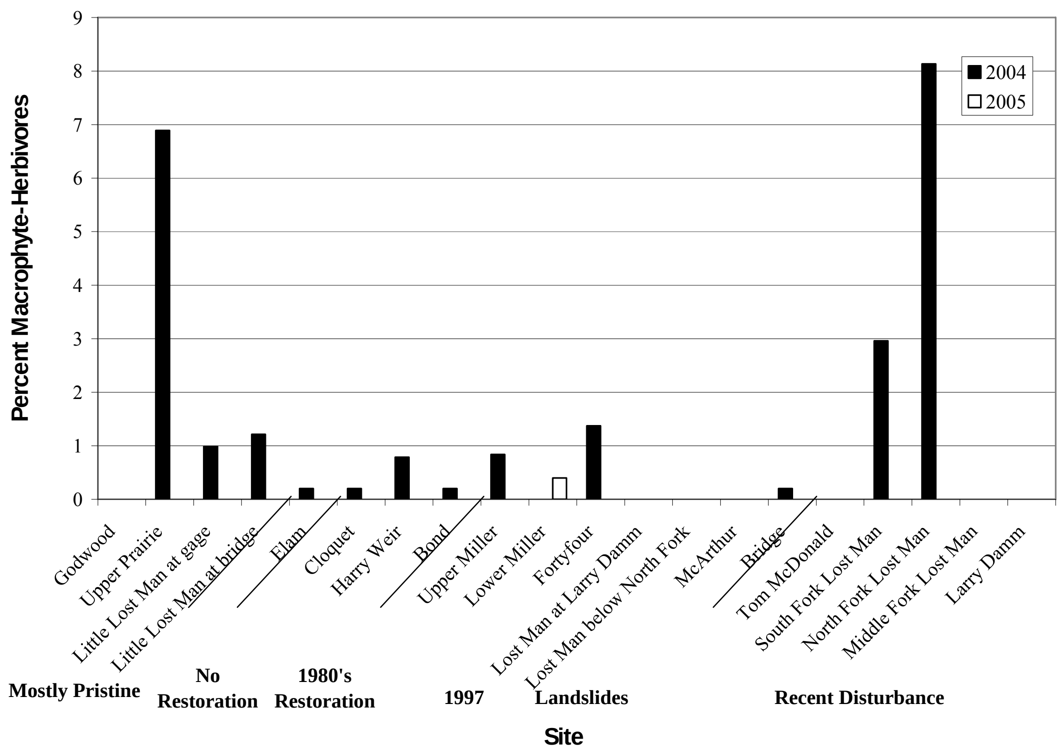
Upper Prairie Creek



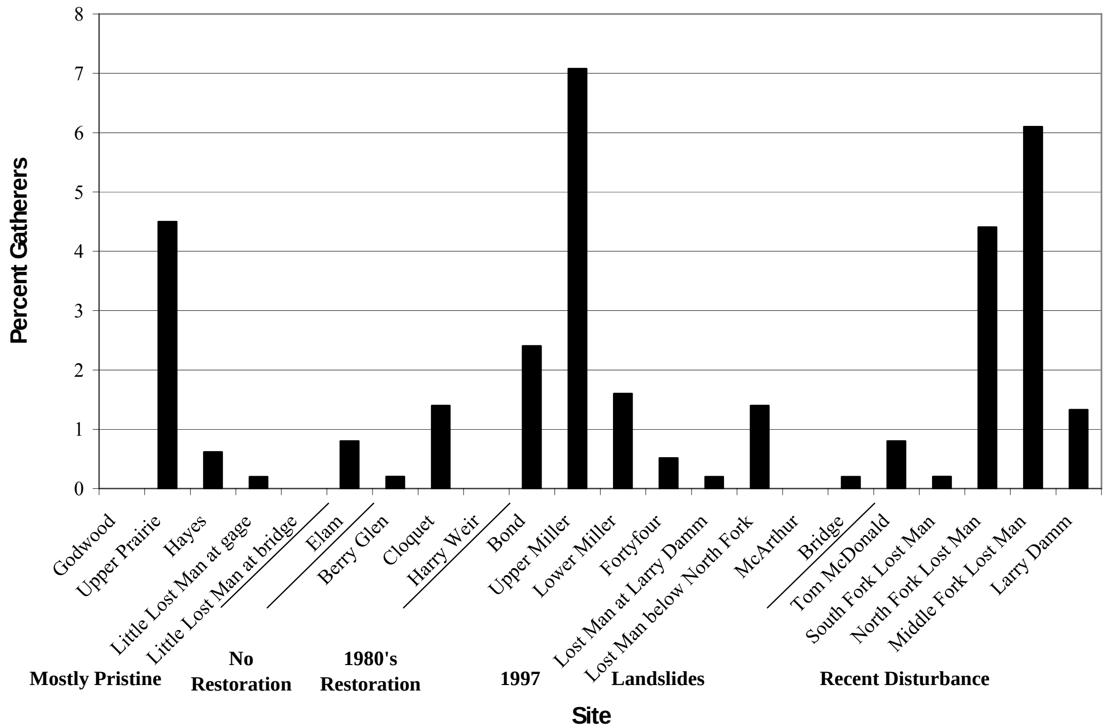
Appendix 4-2u. Functional feeding group percentages for Upper Prairie Creek. GC=gathering collector, FC=filtering collector, SC=scraper, SH=shredder, PR=predator, PH=piercer herbivore.



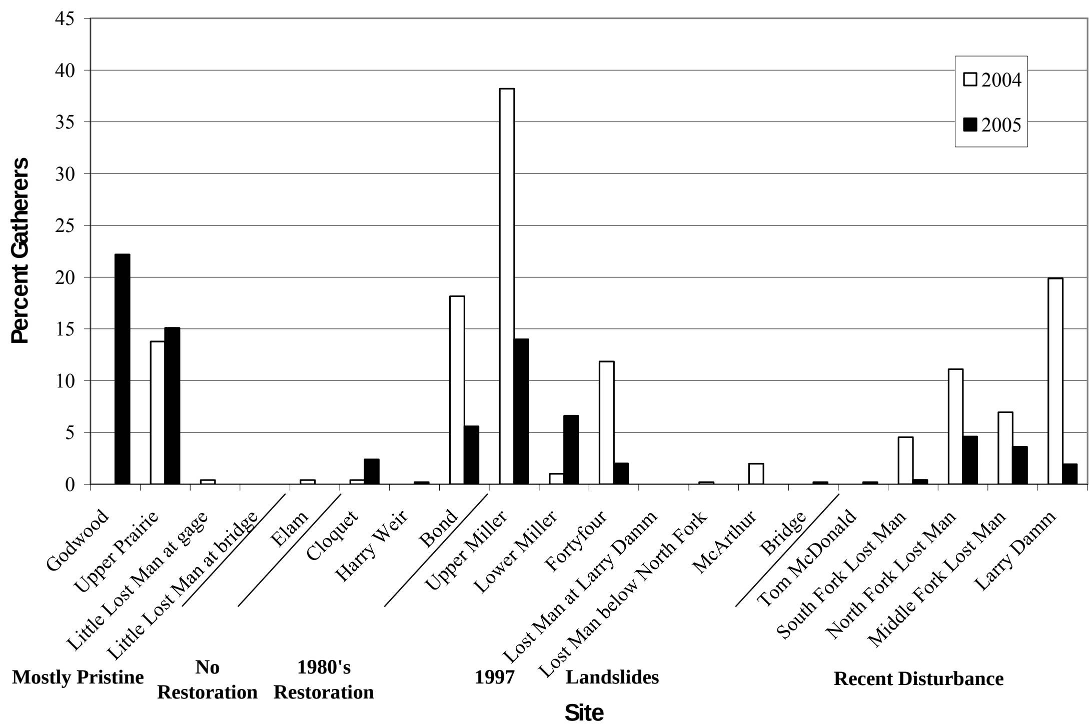
Appendix 4-3. Spring percentage of macrophyte-herbivore macroinvertebrates sampled from tributaries of Redwood Creek in 2005 with a 500μm benthic kick net.



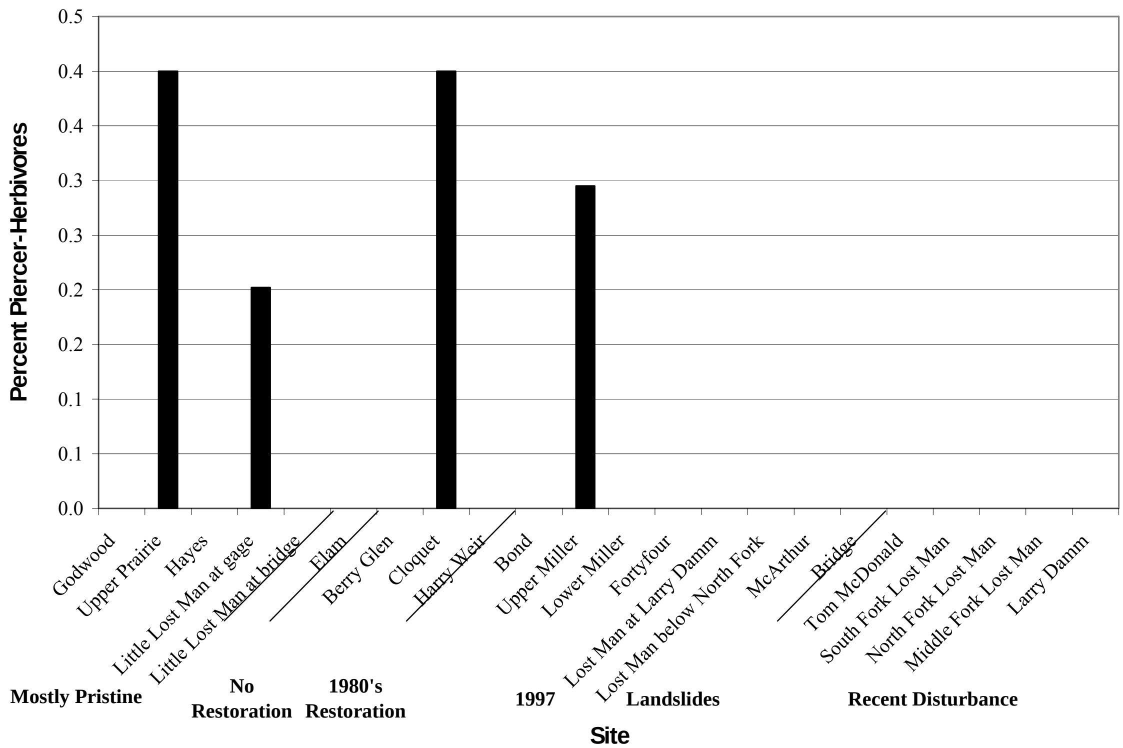
Appendix 4-4. Summer percentage of macrophyte-herbivore macroinvertebrates sampled from tributaries of Redwood Creek in 2004 or 2005 with a 500μm benthic kick net.



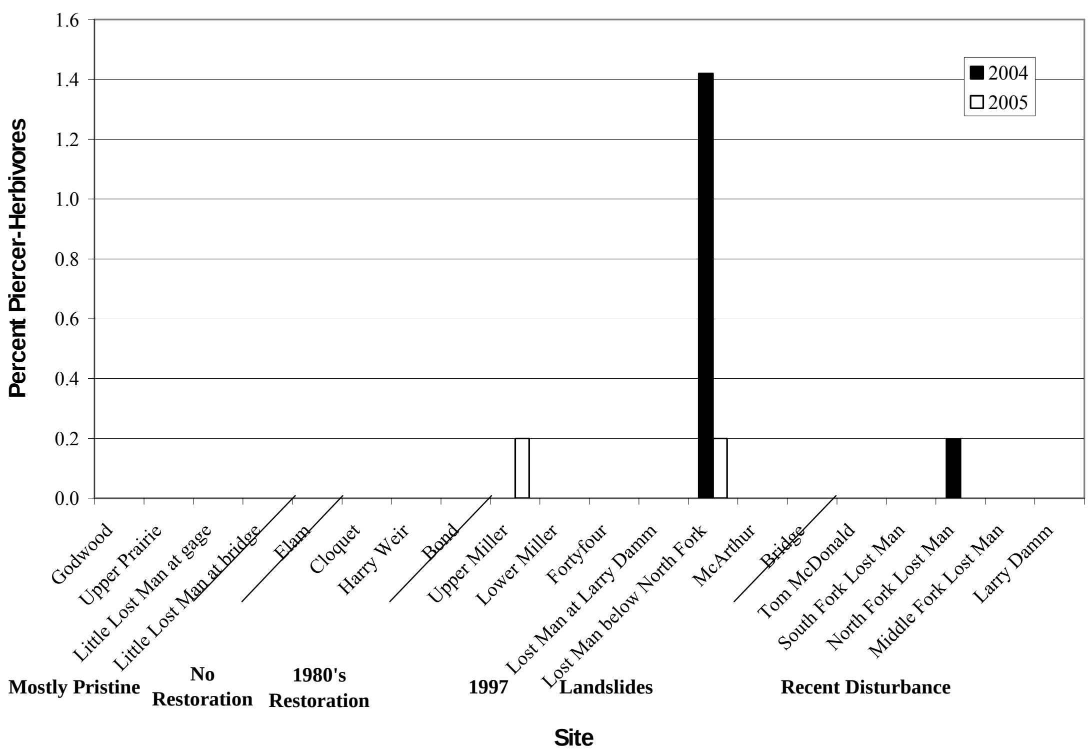
Appendix 4-5. Spring percentage of gathering macroinvertebrates sampled from tributaries of Redwood Creek in 2005 with a 500µm benthic kick net.



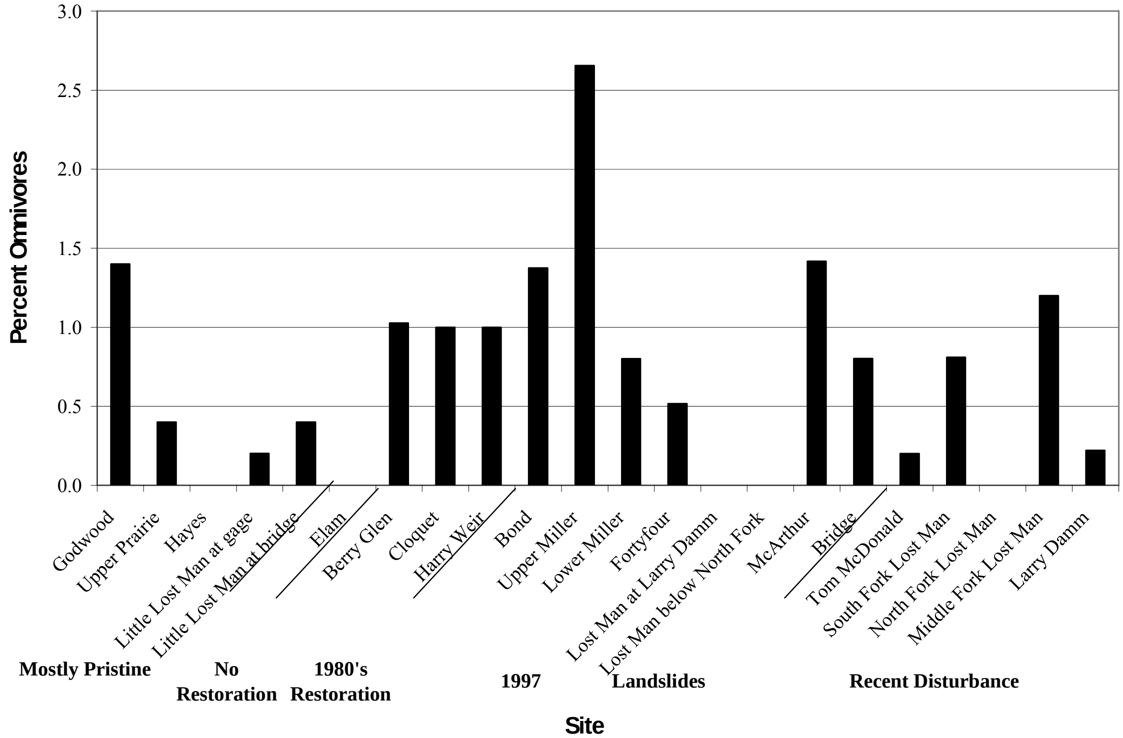
Appendix 4-6. Summer percentage of gathering macroinvertebrates sampled from tributaries of Redwood Creek in 2004 or 2005 with a 500µm benthic kick net.



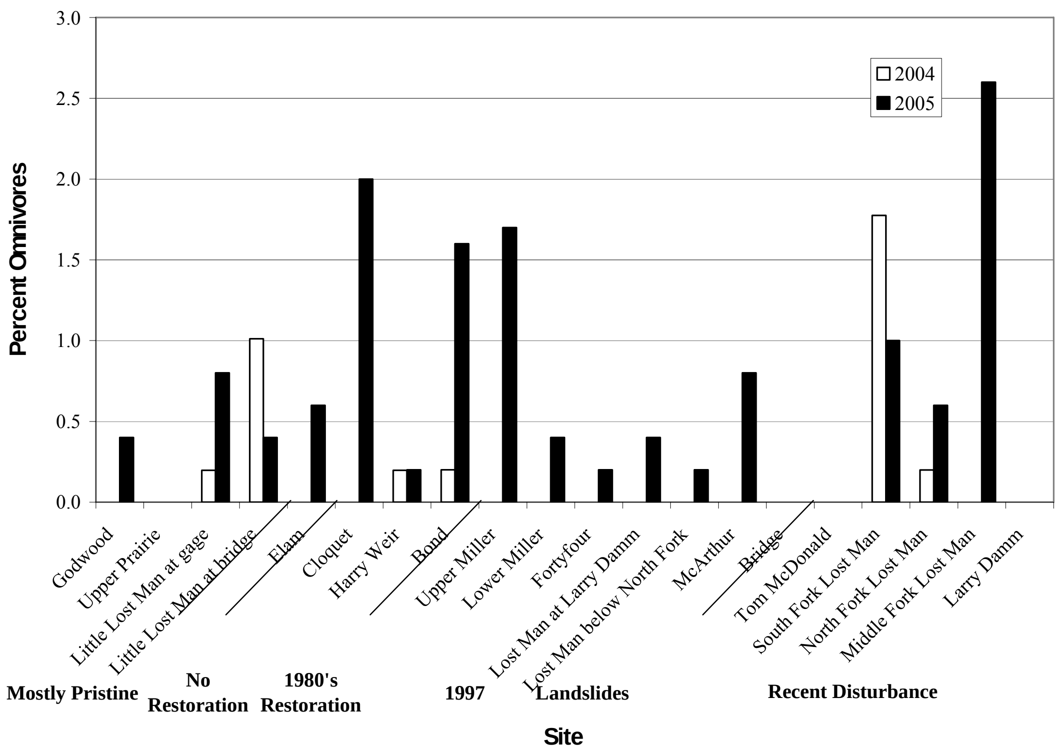
Appendix 4-7. Spring percentage of piercer-herbivore macroinvertebrates sampled from tributaries of Redwood Creek in 2005 with a 500μm benthic kick net.



Appendix 4-8. Summer percentage of piercer-herbivore macroinvertebrates sampled from tributaries of Redwood Creek in 2004 or 2005 with a 500μm benthic kick net.



Appendix 4-9. Spring percentage of omnivore macroinvertebrates sampled from tributaries of Redwood Creek in 2005 with a 500μm benthic kick net.



Appendix 4-10. Summer percentage of omnivore macroinvertebrates sampled from tributaries of Redwood Creek in 2004 or 2005 with a 500μm benthic kick net.

Appendix 6-1. Taxa and number of fish present in tributaries of Redwood Creek over several sampling periods. OK=coho salmon (*Oncorhynchus kisutch*), TR=trout species, OM=steelhead salmon(*Oncorhynchus mykiss*), CUT=cutthroat trout (*Oncorhynchus tshawytscha*), STKL=three spine stickleback (*Gasterosteus aculeatus*), CORA=coast range sculpin (*Cottus aleuticus*), PRICK=prickly sculpin (*Cottus asper*), SCULP=sculpin species, LAMP=brook lamprey (*Lampetra tridentata*), including ammocoetes, CAHU=Humboldt sucker (*Catostomus humboldtianus*), PGS=Pacific giant salamander (*Dicamptodon tenebrosus*), TAILJ=juvenile tailed frog (*Ascaphus trueii*), YELL=adult yellow legged frog (*Rana boylii*), BUFO=adult Western toad (*Bufo boreas*), CRAW=crawfish, RED=adult red legged frog (*Rana aurora draytonii*). Only fish were counted during the 2004 snorkel surveys.

Site	Date	Species	Number of Individuals	Mean Length (mm)	SD	Mean Weight (g)	SD
Harry Weir EMR	5/23/74a	OM	3	81	38	na	na
		CORA	1	na	na	na	na
	7/16/75a	OM	16	39	6	na	na
	9/25/75a	OM	49	53	13	2.2	3.2
		CORA	3	na		na	
		CAHU	1	na		na	
	8/30/94b	OK	27	74	8	4.8	2.0
		OM	43	63	15	3.3	3.2
		PGS	30	35	8	2.0	1.4
	9/2-9/6/04d	OK	0				
		TR	18				
		OM	8				
		PGS	42				
		RED	1				
	7/6/05b	OM	10	69	40	8.4	17.3
		PGS	8	33	8	2.0	1.4
		TAILJ	6	52	3	1.7	0.3
		YELL	1	31	na	3.0	na
	7/14/04*	TR	18				
	7/13/05*	TR	18				

Appendix 6-1 continued.

Site	Date	Species	Number of Individuals	Mean Length (mm)	SD	Mean Weight (g)	SD
Lower Miller	5/24/74a	OK	1	39	na	<1	na
MLRL		OM	2	71	34	na	na
	7/22/75a	OM	7	76	24	na	na
		CORA	3	na	na	na	na
	7/20/04*	TR	13				
	7/18/05*	TR	10				
		PGS	2				
McArthur	11/6/95b	OK	16	82	8	na	na
MCA		OM	24	76	22	na	na
		CUT	2	172	2	na	na
		SCULP	19	67	10	na	na
		PGS	1	130	na	na	na
		TAILJ	10	43	4	na	na
	11/7/95b	OK	19	84	10	8.5	3.1
		OM	22	66	16	4.4	4.3
		SCULP	27	77	16	5.1	-3.9
		PGS	1	31	na	1.2	na
		TAILJ	16	40	3	0.7	0.1
		YELL	1	12	na	0.6	na
	9/1/05b	OK	21	73	5	4.5	1.2
		OM	9	67	18	4.1	3.1
		CUT	1	168	na	48.6	na
		CORA	12	84	13	na	na
		PRICK	6	88	9	na	na
		PGS	4	32	6	na	na
		TAILJ	6	40	4	na	na
		YELL	1	56	na	na	na
	7/12/2004*	OK	55				
		TR	13				

Appendix 6-1 continued.

Site	Date	Species	Number of Individuals	Mean Length (mm)	SD	Mean Weight (g)	SD
	7/19/05*	OK	31				
		TR	8				
		STKL	1				
Larry Damm LDC	8/30/05b	OK	21	64	15	2.9	1.8
		OM	2	52	4	1.6	0.3
		CUT	5	105	35	14.8	10.1
		STKL	1	56	na	na	na
		PRICK	3	116	25	na	na
		SCULP	1	80	na	na	na
		LAMP	1	102	na	na	na
		PGS	18	35	11	na	na
	9/20/05c	OK	28				
		TR	2				
		OC	12				
		PRICK	2				
		LAMP	1				
		PGS	21				
	7/6/2004*	OK	22				
		TR	11				
	7/15/05*	OK	34				
		TR	9				
		STKL	1				
Elam ELA	9/1/99d*	OK	2				
		TR	37				
		PGS	10				
		TAILJ	4				
		TAILA	1				

Appendix 6-1 continued.

Site	Date	Species	Number of Individuals	Mean Length (mm)	SD	Mean Weight (g)	SD
	9/1/04d	OK	50				
		TR	43				
		OM	1				
		STKL	1				
		CORA	1				
		PGS	37				
		TAILJ	4				
	9/1/05b	OK	19	67	8	3.9	1.5
		OM	11	67	18	4.5	3.6
		CUT	1	183	na	70.1	na
		CORA	11	81	12	na	na
		PRICK	4	86	16	na	na
		PGS	5	na	na	na	na
		TAILJ	19	33	3	na	na
	7/13/04*	OK	82				
		TR	34				
	7/19/05*	OK	72				
		TR	8				
		TAILJ	6				
Little Lost Man at Bridge LLML	5/22/74a	OK	10	53	11	na	na
		OM	19	96	96	12.6	13.1
		STKL	10	na	na	na	na
	7/23/75a	OM	72	70	20	na	na
		CORA	5	na	na	na	na
		STKL	7	na	na	na	na
	9/24/75a	OM	39	86	34	11.7	23.5
		STKL	observed only				
		LAMP	observed only				

Appendix 6-1 continued.

Site	Date	Species	Number of Individuals	Mean Length (mm)	SD	Mean Weight (g)	SD
	7/7/05b	OK	26	57	6	2.1	0.6
		OM	21	50	23	2.5	4.4
		LAMP	1	135	na	na	na
		PGS	13	62	27	na	na
		TAILJ	18	46	3	1.3	0.4
	7/1/04*	OK	99				
		TR	15				
		STKL	12				
	7/15/05*	OK	81				
		TR	36				
Bridge BRI	5/23/74a	CORA	3	na	na	na	na
	7/17/75a	OM	16	60	37	na	na
	9/25/75a	OM	48	68	16	4.8	3.9
		CORA	11	na	na	na	na
		CAHU	1	na	na	na	na
	9/10/96b	OM	171	79	29	8.2	11.7
		PGS	26	61	24	12.0	16.2
		TAILJ	3	44	2	0.8	0.1
		BUFO	1	31	na	4.0	na
	8/24/01d*	TR	10				
		OM	98				
		OC	49				
		CORA	2				
		PRICK	2				
		PGS	19				
	9/19-9/26/02d*	OK	170				
		TR	214				
		OM	721				

Appendix 6-1 continued.

Site	Date	Species	Number of Individuals	Mean Length (mm)	SD	Mean Weight (g)	SD
	9/10-9/22/03d*	OK	28				
		TR	926				
		LAMP	6				
		PGS	181				
		TAILJ	5				
	10/4-10/12/04d*	OK	5				
		TR	630				
		OM	295				
		CORA	28				
		LAMP	3				
		PGS	152				
		YELL	1				
	7/6/05b	OK	2	62	6	2.8	0.9
		OM	2	73	46	6.6	8.6
		CORA	10	69	6	3.4	0.9
		PRICK	3	91	15	8.4	4.0
		SCULP	1	78	na	5.3	na
		LAMP	1	148	na	6.1	na
	7/8/04*	OK	31				
		TR	149				
	7/13/05*	OK	11				
		TR	49				
Tom McDonald TMC	5/24/74a	OK	15	39	6	<1	na
		STKL	1	na	na	na	na
		CORA	3	na	na	na	na
	7/21/75a	OM	44	57	38	na	na
		CORA	19	na	na	na	na
		STKL	5	na	na	na	na

Appendix 6-1 continued.

Site	Date	Species	Number of Individuals	Mean Length (mm)	SD	Mean Weight (g)	SD
	9/25/75a	OM	50	69	22	4.7	6.9
		CORA	6	na		na	
		LAMP	observed only				
	8/23/94b	OK	6	67	4	3.2	1.0
		OM	18	71	20	4.9	4.9
		PGS	5	51	17	7.2	6.9
	7/7/05b	OK	12	58	8	2.3	1.2
		OM	8	41	8	0.6	0.3
		CORA	8	82	11	6.9	2.2
		PRICK	1	130	na	30.4	na
		PGS	2	53	19	8.3	7.4
		TAILJ	16	45	3	1.0	0.2
		YELL	1	60	na	na	na
	7/7/04*	OK	7				
		TR	18				
	7/13/05*	OK	33				
		TR	3				
Lost Man below North Fork LM3	10/17/95b	OK	61	69	8	4.4	1.7
		OM	84	58	17	3.0	4.8
		CUT	4	131	23	24.7	15.1
		LAMP	4	112	29	2.7	1.9
		PGS	19	54	34	14.5	28.7
	10/18/95b	OK	61	70	9	6.0	9.8
		OM	109	64	18	4.3	5.8
		CUT	3	102	33	13.0	11.7
		LAMP	6	122	4	2.8	0.5
		PGS	15	67	39	20.1	28.1
		TAILJ	1	42	na	0.9	na
	7/6/04*	OK	85				
		TR	64				

Appendix 6-1 continued**.**

Site	Date	Species	Number of Individuals	Mean Length (mm)	SD	Mean Weight (g)	SD
	7/21/05*	OK	73				
		TR	64				
North Fork Lost Man NFL	8/29/05b	OK	5	63	10	3.3	1.4
		OM	4	52	6	1.7	0.5
		CUT	7	88	27	8.5	6.7
		PGS	10	36	11	na	na
		TAILJ	9	38	2	na	na
		BUFO	1	43	na	na	na
	9/13/2005c	AM	2				
		AT	18				
		DT	30				
		CUT	9				
		OK	23				
		OM	1	83	na	6.2	na
		TR	19				
	7/27/04*	OK	31				
		TR	13				
		OM	2				
		CUT	4				
	7/21/05*	OK	32				
		TR	23				
Hayes HAY	9/12/96b	OM	25	64	14	3.3	2.4
		CUT	5	117	29	19.9	13.1
		PGS	27	42	24	6.4	15.7
		YELL	5	36	4	4.7	1.5
	7/12/04*	TR	1				
		CUT	6				
	7/19/05*	PGS	7				
		TAILJ	1				

Appendix 6-1 continued.

Site	Date	Species	Number of Individuals	Mean Length (mm)	SD	Mean Weight (g)	SD
Fortyfour FOR	10/4/95b	OM	3	67	3	3.5	0.6
		CUT	15	131	33	26.3	17.4
		PGS	28	47	24	11.0	22.2
	7/15/2004*	TR	7				
	7/14/05*	TR	19				
Little Lost Man at Gage LLM	9/26/94b	OM	49	59	22	3.6	4.6
		PGS	22	49	19	6.8	11.6
		TAILJ	3	50	3	1.5	0.3
	9/27/94b	OM	28	63	24	4.2	6.4
		PGS	15	64	31	17.9	26.1
		TAILJ	45	49	2	1.4	0.1
	9/21/05c	OK	52	93	17	9.8	4.5
		TR	78				
		OM	7				
		OC	3				
		DT	50				
	7/1/04*	OK	13				
		TR	87				
	7/21/05*	OK	61				
		TR	48				
		TAILJ	1				
Upper Prairie PRU	6/30/04*	OK	47				
		TR	21				
		STKL	1				
	7/12/05*	OK	36				
		TR	23				
		STKL	1				

Appendix 6-1 continued.

Site	Date	Species	Number of Individuals	Mean Length (mm)	SD	Mean Weight (g)	SD
Lost Man at hatchery	8/29/05[b]	OK	26	65	8	3.2	1.1
		CORA	5	89	46	na	na
		PGS	3	70	61	na	na
		PRICK	14	96	23	na	na
		SCULP	2	71	13	na	na
		OM	14	79	29	7.7	8.8
		STKL	6	54	2	na	na
	9/18/05[c]	OK	392	na		na	
		TR	97	61	4	2.7	0.5
		OM	50	91	22	10.0	8.1
		CUT	5	na		na	
		STKL	41				
		PRICK	51				
		LAMP	55				
		PGS	36				
		CORA	49				
		CRAW	3				
	9/19/05[c]	OK	112				
		TR	50				
		OM	17	85	18	8.2	6.4
		CUT	3				
		STKL	3				
		PRICK	28				
		LAMP	7				
		PGS	64				
		CORA	42				
		CRAW	1				
Lost Man above Larry Damm LM2	7/19/04*	OK	69				
		TR	94				
		STKL	4				
	7/15/05*	OK	92				
		TR	27				

Appendix 6-1 continued.

Site	Date	Species	Number of Individuals	Mean Length (mm)	SD	Mean Weight (g)	SD
Bond	7/20/04*	TR	15				
BON							
	7/14/05*	TR	7				
Cloquet	7/21/04*	TR	9				
CLO							
	7/18/05*	OK	2				
		TR	2				
		PGS	9				
South Fork	7/22/04*	TR	81				
Lost Man							
SFL	7/15/05*	TR	20				
Middle Fork	7/22/04*	TR	6				
Lost Man		OM	19				
MFL		CUT	21				
	7/15/05*	TR	52				
		PGS	1				
		TAILJ	1				
Upper Miller	7/28/04*	none					
MLRU							
	7/21/05*	PGS	6				
		TAILJ	1				
Godwood	9/23/2005c	OK	29				
GOD		TR	19				
		CUT	4				
		STKL	2				
		PRICK	3				
		LAMP	2				
		PGS	23				
	7/12/05*	OK	26				
		TR	18				
		STKL	1				

Appendix 6-1 continued.

Site	Date	Species	Number of Individuals	Mean Length (mm)	SD	Mean Weight (g)	SD
Little Lost Man	9/26/00d*	OK	7				
		TR	98				
		PGS	20				
		TAILJ	1				
Lost Man	8/11-8/13/01d*	OK	265				
		TR	357				
		OM	27				
		OC	10				
		STKL	69				
		LAMP	32				
		PGS	20				
		CRAW	4				
	9/24-10/2/03d*	OK	346				
		TR	900				
		OC	1				
		STKL	13				
		LAMP	17				
		PGS	215				
		TAILA	1				
		CRAW	1				

* = data collected from snorkel surveys; all other data collected from electrofishing
a = data collected by Iwatsubo and Averett
b = data collected by Redwood National and State Parks fishery biologists
d = data collected by Dana McCanne